A COMPARISON OF OLDGROWTH MIXED FOREST WITH REGENERATION RESULTING FROM LOGGING OR WILDFIRE

by

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DECLARATION

This thesis contains no material which has been accepted for the award of any other degree or diploma in any university, and to the best of my belief, contains no copy or paraphrase of material previously published or written by another person, except where due reference is made in the text of the thesis.

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ABSTRACT

Up to 194,000 ha, or 20 percent, of Tasmania's wet eucalypt forest is mature mixed forest greater than 110 years old. At least 33 percent of the mixed forest is reserved in the Tasmanian Wilderness World Heritage Area (TWWHA) or in State and Forest Reserves outside the TWWHA. Almost half the area of mixed forest with a mature myrtle understorey has a eucalypt density of only 5-20 percent which implies that it is in the last successional stage prior to becoming rainforest.

A comparison of the floristics of 20-30-year-old silvicultural and wildfire regeneration with oldgrowth mixed forest showed that species richness for all lifeforms other than epiphytic ferns was greater in regenerated forest than in oldgrowth mixed forest. Most of the common species in oldgrowth mixed forest were represented in approximately similar frequencies in silvicultural and wildfire regeneration. The major floristic difference between the two regeneration types was the much lower frequency of epiphytic fern species in silvicultural regeneration. The floristic composition of oldgrowth mixed forest is profoundly influenced by either wildfire or clearfelling and slash burning but the composition of the regeneration is more influenced by environmental variables than by the nature of the disturbance which initiated the regeneration.

Comparisons indicated only slight differences in growth and density of tree and tall shrub species between silvicultural and wildfire regeneration. Growth of rainforest tree species in young regenerated stands was very slow due, apparently, to suppression by a dense layer of taller sclerophyllous trees and shrubs. The density of most rainforest tree species was lower in regenerated sites than in oldgrowth mixed forest but there was no significant difference between logged and wildfire sites. The density of common rainforest shrub species was similar in all three site types.

Several rainforest species with short seed dispersal distances can regenerate vegetatively after fire. There was no difference between silvicultural and wildfire regeneration in the amount of vegetative regeneration of rainforest canopy species. Basal sprouting of rainforest canopy species occurred in the first year following disturbance but seedlings took several years to become established. No significant differences were found in seedling establishment time of rainforest canopy species. Five woody rainforest species regenerated from soil seed banks and included two species which took more than one year to germinate. The species similarity between oldgrowth mixed forest stands and their soil seed banks was low.

Coupe sizes are much too large for *Nothofagus cunninghamii* seed sources from adjacent stands to regenerate the logged area within the planned rotation age. The frequently observed regeneration of this species within coupes must occur from vegetative sources, seeds and seedlings which have survived the regeneration burn, or seed that is shed subsequently by surviving mature trees.

An analysis of data from randomly located permanent plots in eucalypt regeneration established on mixed forest and rainforest sites showed that 57 percent of sites contained *Nothofagus cunninghamii* in the regeneration at 20-30 years of age. There was no significant difference in the presence of *N. cunninghamii* between stands which had been regenerated by artificially sowing eucalypt seed, stands regenerated by seedfall from retained eucalypt seedtrees, and stands regenerated by a wildfire.

Mean diameter increments of rainforest timber species in regrowth forests were only two to four mm per year. Long rotations, perhaps around 200 years, would be needed for most rainforest timber species to reach millable sizes.

Mixed forest used for wood production is logged on rotations of 80-100 years which is an insufficient time for the redevelopment of mature mixed forest. Alternative silvicultural methods, and longer rotations, need to be considered for the treatment of mixed forest used for wood production if the conservation and special timber values of mixed forest are to be maintained. Nine silvicultural treatments for mixed forest are listed. The critical factor in the silvicultural perpetuation of mixed forest may be rotation length rather than regeneration treatment.

CONTENTS

FIGURES

1. INTRODUCTION

Temperate evergreen forests occur in widely disjunct areas of the world which include South America, New Zealand, Australia, southeastern North America, the Middle East, India, Africa and Japan (Ovington 1983). Although they differ greatly in the species composition of their flora they show a general affinity in structure and a tendency for the tree species to be predominantly *Quercus* in the Northern Hemisphere and *Nothofagus* or *Eucalyptus* in the Southern Hemisphere (*ibid*). They can be broadly divided into temperate broad-leaved sclerophyll forest and temperate broad-leaved rainforest. Ashton (1981[a]) notes that in some places in Australia the two types occur together as mature wet sclerophyll eucalypt emergents over a well developed rainforest. This forest-in-forest structure is successional as the rainforest species are self-replacing while the eucalypts are light-demanding and require open conditions for seedling establishment (*ibid*). Such stands were described by Gilbert (1959) as mixed forests.

The term mixed forest is applied mainly to the extensive transitional forest which occurs, either spatially or temporally, between rainforest and wet sclerophyll forest in southeastern Australia. In northern Australia, the boundaries between rainforest and sclerophyll vegetation are much sharper (Adam 1992) and the term has little application. Internationally, the term mixed forest is often used in a general sense to describe broadly a mixture of forest types, e.g. mixed oak forests, mixed eucalypt forests, mixed conifer forests and mixed hardwood forests. There appears to be little consistency in the forest types being described except where reference is made to the mixed forests of southeastern Australia.

Lowland mixed forest (*sensu* Gilbert 1959) is maintained by infrequent wildfires with an interval of about 100 to 350 years (Gilbert 1959, Jackson 1968, Mount 1979) which allow the re-establishment of eucalypts and wet sclerophyll understorey species. If fires are too infrequent (i.e. at intervals exceeding about 400 years) the eucalypts die without being replaced and the forest becomes rainforest.

Mixed forests with a cool temperate rainforest understorey have their greatest extent and diversity in Tasmania. Kirkpatrick *et al.* (1988) recognised 37 mixed forest communities, including 12 subalpine communities, in their classification of Tasmanian wet eucalypt communities.

Mixed forests have high value for wood production and are an important source of eucalypt and rainforest species veneer and sawlogs. The non-eucalypt timber species are frequently referred to as special timbers (e.g. Forestry Commission 1990[a]). The major special timbers derived from mixed forest are blackwood (*Acacia melanoxylon*), celery-top pine (*Phyllocladus aspleniifolius*), leatherwood (*Eucryphia lucida*), myrtle (*Nothofagus cunninghamii*), sassafras (*Atherosperma moschatum*) and silver wattle (*Acacia dealbata*). These are rainforest species (*sensu* Jarman and Brown 1983) except for blackwood and silver wattle which are classed as 'doubtful' rainforest species (Jarman *et al.* 1991). Mesibov (1983) has listed a further 27 non-eucalypt tree species which can be used for craftwood including 13 species common in mixed forest. Mixed forests with a high proportion of leatherwood trees are also economically important as sources of nectar for apiculture (Neyland and Hickey 1990).

The standard silvicultural technique for regeneration of lowland mixed forest is to clearfell, burn and broadcast sow with eucalypt seed (Gilbert and Cunningham 1972). The notional rotation age for wet eucalypt forests is 80 to 100 years although small areas are sometimes assigned to longer rotation cycles primarily to conserve patches of tall forest for recreational and aesthetic purposes (Hickey and Brown 1989). Duncan (1985) has summarised the effects of forest practices on mixed forest and concluded from the studies of Cunningham and Cremer (1965) and Taplin (1982) that the logging of regrowth forests on a rotation shorter than the period required for re-establishment of mature mixed forest will lead to a shift towards a wet sclerophyll understorey on those sites which have a sclerophyll seed source. This has led to concern from conservationists (e.g. Combined Environment Groups 1988), special timber users (e.g. de Lara 1988), apiarists (e.g. Hoskinson and Graeme-Evans 1988) as well as scientists (e.g. Brown *et al.* 1988, Barker 1992) that many of the values of mixed forests, particularly those associated with rainforest species, are being compromised by their management for eucalypt sawlogs on rotations of 80 to 100 years.

This study aimed to:

- determine the extent and reservation status of mixed forests;
- compare the floristics of oldgrowth mixed forest with those of 20-30-year-old regeneration resulting from logging or wildfire;
- compare the structure of silvicultural and wildfire regeneration;

- report the mechanisms for rainforest species regeneration following massive disturbance;

- analyse data on growth of special timbers species in regrowth eucalypt forest; and

- consider options to maintain the conservation and special timber values of mixed forest used for wood production.

The definition of oldgrowth used in this study is that used for mapping purposes by the Forestry Commission, Tasmania (see Forests and Forest Industry Council 1990) and includes eucalypt forest more than 110 years old. There are other definitions of oldgrowth (see Resource Assessment Commission 1992) but it is likely that oldgrowth sites described in this study would be similarly categorised under the alternative definitions. The nomenclature of species follows Buchanan *et al.* (1989).

2. EXTENT, DISTRIBUTION AND TENURE

The extent of mixed forests in Tasmania has not been reported although it is primarily a subset of the 994,000 ha of wet eucalypt forest shown on the 1:500,000 *Forests of Tasmania* map (Forestry Commission 1990[b]). The area of mixed forest can be determined at a local level from large-scale photo-interpreted (PI) maps prepared by the Forestry Commission. These maps divide eucalypt forests into height and crown cover classes, shown in Table 2.1, for mature (defined as older than 110 years) and regrowth (less than or equal to 110 years) stands. Heights of stands are estimated by skilled photo-interpreters. Heights of some individual trees are measured using a parallax device in order to validate ocular judgements. Spurr (1948) indicates that an average observer should be able to classify stands into 10 foot height classes using this method while skilled interpreters can achieve more precise results. Density classes are a measure of crown cover, i.e. the proportion of each stand that is covered by the crowns of the eucalypt overstorey. The crowns are assumed to be solid. This methodology differs from many ground-based forest classifications where projective crown cover is used (e.g. Specht 1970). Some examples of legends from archived and current PI maps are shown as Appendix 1.

Rainforests are broadly denoted on PI maps by the code 'M' which indicates forest containing myrtle (*Nothofagus cunninghamii*). Other non-eucalypt tree species, referred to as 'secondary species' on PI maps, may be indicated by the code 'T'. This can represent rainforest tree species such as *Atherosperma moschatum*, *Eucryphia lucida* and *Phyllocladus aspleniifolius*. Equally, it may indicate non-rainforest species such as tea-tree (*Leptospermum* and *Melaleuca* spp.), silver wattle (*Acacia dealbata*), blackwood (*A. melanoxylon*), lancewood (*Phebalium squameum*) and native pear (*Pomaderris apetala*). Since 1986 the rainforest typing specifications have been simplified (Hickey *et al.* 1993) so that rainforests and rainforest understoreys are shown as:

- M+ which represents rainforest which is usually taller than 25 m and occurs on more fertile sites; or
- M- which refers to rainforest from 8-25 m tall which occurs on sites of moderate to low fertility.

Table 2.1 Eucalypt height and density classes shown on Forestry Commission PI maps

Height classes

Mature eucalypt		Regrowth eucalypt	
E1*	> 76m	ER6	> 50m
E1	55-76m	ER5	44-50m
E2	41-55m	ER4	37-44m
E3+	34-41m	ER3	27-37m
E3-	27-34m	ER2	15-27m
E4	15-27m	ER1	< 15m
E5	< 15m		

Density (percent crown cover) classes

class	Mature eucalypt	Regrowth eucalypt
a	70-100%	90-100%
b	40-70%	70-90%
c	20-40%	50-70%
d	5-20%	10-50%
f	< 5%	1-10%

Mixed forests are represented by PI types which indicate a eucalypt overstorey and a rainforest understorey, e.g. E2bMT (pre-1986) or its equivalent, E2bM+ (since 1986). PI types such as these always indicate mature mixed forest. PI types without M but with T may represent mixed forest but equally may represent a forest with a tall sclerophyllous understorey. Types which do not indicate any understorey , such as E2b, may include forests with immature rainforest trees which are not discernible on aerial photography. By convention, the Forestry Commission considers vegetation with a eucalypt overstorey and a rainforest understorey to be rainforest if the eucalypt crown cover is less than five percent, i.e. if it has a mature eucalypt density of 'f'.

By 1991 PI maps were available, in digital form, for all mixed forest and rainforest areas in Tasmania. The date of photography used to compile the maps is shown in Fig. 2.1. The Forestry Commission's inventory system known as FORAST is used to store and process areal information about native forests.

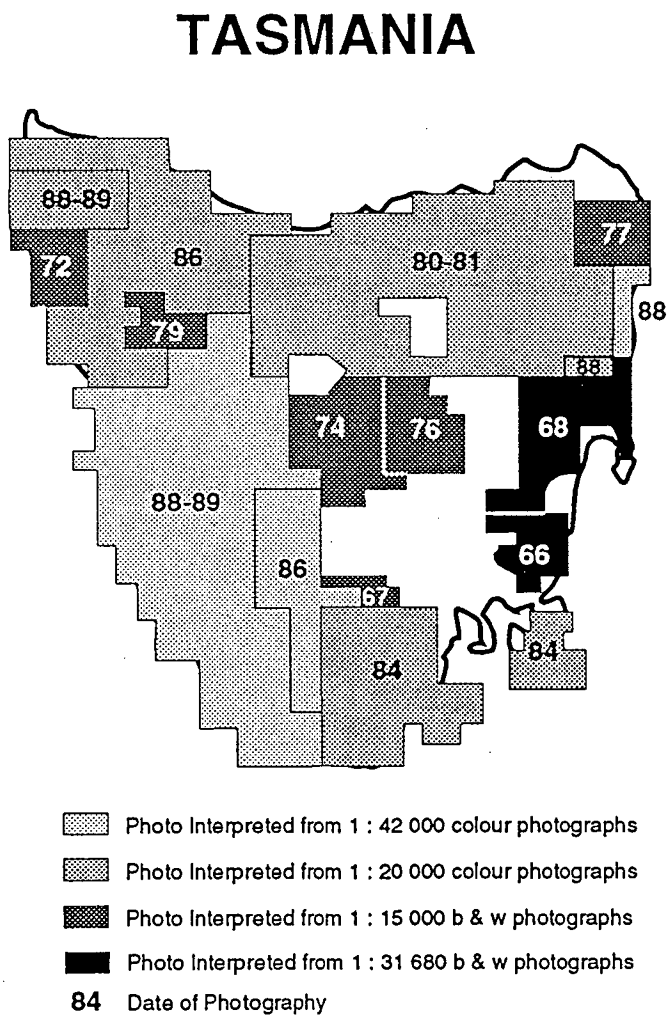
Method

The FORAST database was used to determine the extent, tenure and approximate distribution of mixed forest in Tasmania. The tenure classification was taken from edition 4 of the 1:500,000 land map of Tasmania (Department of Environment and Planning 1990). The area of mixed forest with mature myrtle was obtained from FORAST by tallying PI types which contained M and the following mature eucalypt density classes: a-c, d and f. The d and f density classes were tallied separately as the former represents the mixed forest class which is most akin to rainforest while the latter is significant because, although it is classed by the Forestry Commission as rainforest, it has presumably been subjected to fire or some other major disturbance within the last 400 years. PI types which included both mature and regrowth eucalypts with d density and a rainforest understorey were classed as a-c density. However, types which had mature and regrowth eucalypts of f density were tallied as f density.

The area of eucalypt forest with an understorey of secondary species excluding mature myrtle was obtained by tallying eucalypt types which contained T but not M. All the areas were tallied according to their respective 1:25,000 map sheets.

The production of an accurate small-scale map of mixed forest in Tasmania was beyond the scope of this project but an approximation was prepared by representing areas of mixed forest on 1:25,000 map sheets as shaded circles at the centroids of each sheet. The area of each circle represents the area of mixed forest on each map sheet.

Fig. 2.1 Date of aerial photography used to compile estimates of the extent of mixed forest.



Results

The area and tenure of mixed forest in Tasmania with at least 5 percent eucalypt crown cover (i.e. mature eucalypt density classes a, b, c and d), and a mature myrtle understorey, is shown in Table 2.2. This table indicates there are about 112,900 ha of mixed forest with a mature myrtle understorey in Tasmania of which 43 percent occurs in the Tasmanian Wilderness World Heritage Area (TWWHA) or in State Reserves outside the TWWHA. Fifty-two percent of the mixed forest has a potential eucalypt height of at least 41 m while only 2 percent has a potential eucalypt height of less than 27 m. There are about 51,600 ha of mixed forest on State forest or unallocated Crown land. Fig. 2. 2 shows the approximate distribution of mixed forest in Tasmania with at least 5 percent eucalypt crown cover, and a mature myrtle understorey.

The area of mixed forest with a sparse (d density) eucalypt overstorey is shown in Table 2.3 and the area of rainforest with a very sparse (f density) eucalypt overstorey is shown in Table 2.4. Table 2.3 shows that almost half the total forest area in Table 2.2 has a sparse eucalypt crown cover of 5-20 percent. Table 2.4 indicates there are 46,700 ha of rainforest (or about 6 percent of the total area of rainforest in Tasmania) which have a very sparse eucalypt crown cover. Forty-nine and 46 percent of the total areas shown in Tables 1.3 and 1.4 respectively occur in the TWWHA or in State Reserves outside the TWWHA.

Table 2.5 shows the area and tenure of eucalypt forest in Tasmania with at least 5 percent eucalypt crown cover and an understorey of secondary species. There are about 81,300 ha of eucalypt forest with an understorey of secondary species which may include rainforest trees. The proportion of this forest type which is mixed forest and the proportion which is wet sclerophyll forest cannot be determined from PI type maps. However, by considering the total area figures from Tables 2.2 and 2.5 it can be deduced that the area of mature mixed forest in Tasmania lies between 113,000 and 194,000 ha. This represents 11-20 percent of the total area of wet eucalypt forest. At least 33 percent of the mature mixed forest is in formal reserves, i.e. World Heritage Area, State Reserves and Forest Reserves.

Table 2.2. Area and tenure of mixed forest with at least 5 percent eucalypt crown cover and a mature myrtle understorey. (The E3 height class refers to PI types which have not been subdivided into E3+ and E3-.)

	E1	E2	E3	E3+	E3-	E4	E5	TOTAL	%
	ha	ha	ha	ha	ha	ha	ha		
TWWHA	1,988	13,162	899	15,721	14,000	1,452	0	47,222	42
Other State Reserves (National Parks)[1]	114	187	2	448	419	59	0	1,229	1
Forest Reserves1	56	173	18	83	10	0	0	340	<1
Crown land in Conservation Areas[2]	113	874	0	4,810	1,620	188	0	7,605	7
Protected Areas	0	31	0	2	0	30	0	63	<1
State Forests[1]	7,164	23,420	4,828	2,617	1,245	162	1	39,437	35
Unallocated Crown Land	163	7,830	166	3,178	706	103	33	12,179	11
HEC-vested land	0	33	0	150	160	31	0	374	<1
Other Crown Reserves	0	8	0	1	0	0	0	9	<1
Private property	49	2,256	13	1,813	281	0	0	4,412	4
TOTAL	9,647	47,974	5,926	28,823	18,441	2,025	34	112,870	100
PERCENT	9	43	5	25	16	2	<1	100	

[1] outside the TWWHA

[2] under the sole control of the Department of Environment and Land Management

Fig. 2.2 The distribution of mixed forest with at least five percent eucalypt (a-d density) crown cover, and a mature myrtle understorey.

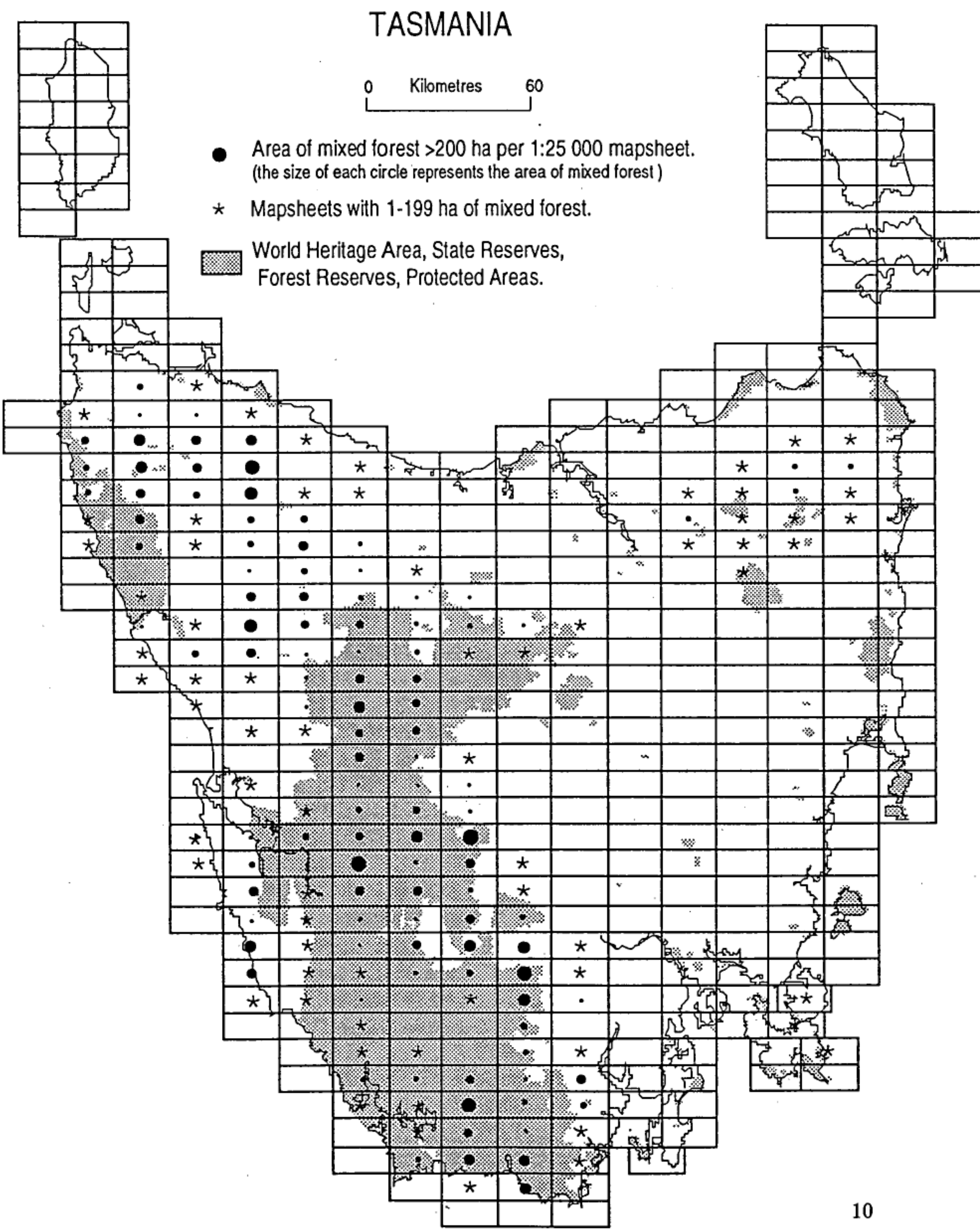


Table 2.3. Area and tenure of mixed forest with a sparse (5-20 percent) eucalypt cover and a mature myrtle understorey.

	E1d	E2d	E3d	E+3d	E-3d	E4d	E5d	TOTAL	%
	ha	ha	ha	ha	ha	ha	ha	ha	
TWWHA	925	6,238	501	7,700	9,811	1,274	0	26,449	48
Other State Reserves (National Parks)	2	108	0	210	419	27	0	766	<1
Forest Reserves	23	74	12	50	10	0	0	169	<1
Crown land in Conservation Areas	65	361	0	1,715	950	188	0	3,279	6
Protected Areas	0	7	0	2	0	30	0	39	<1
State Forests	2,264	9,121	1,332	1,894	888	92	0	15,951	29
Unallocated Crown Land	78	2,562	18	1,378	539	101	13	4,689	9
HEC-vested land	0	23	0	66	73	31	0	193	<1
Other Crown Reserves	0	8	0	1	0	0	0	9	<1
Private property	48	1,748	13	1,359	223	0	0	3,391	6
TOTAL	3,765	20,250	1,876	14,375	12,913	1,743	13	54,935	100
PERCENT	7	37	3	26	24	3	<1	100	

Table 2.4. Area and tenure of rainforest with a very sparse (less than 5 percent) eucalypt overstorey.

	E1f	E2f	E3f	E+3f	E-3f	E4f	E5f	TOTAL	%
	ha	ha	ha	ha	ha	ha	ha	ha	
TWWHA	208	7,274	348	6,436	6,577	357	0	21,201	45
Other State Reserves (National Parks)	61	546	0	4	46	0	0	657	1
Forest Reserves	6	96	0	34	0	0	0	136	<1
Crown land in Conservation Areas	0	308	0	546	315	5	0	1,175	3
Protected Areas	7	6	0	0	35	3	0	51	<1
State Forests	2,000	10,779	536	1,230	924	58	0	15,524	33
Unallocated Crown Land	7	5,072	47	667	349	19	0	6,160	13
HEC-vested land	0	70	0	20	16	0	0	106	<1
Other Crown Reserves	2	5	0	24	0	0	0	30	<1
Private property	57	1,156	6	348	54	0	0	1,621	4
TOTAL	2,348	25,312	937	9,309	8,316	442	0	46,661	100
PERCENT	5	54	2	20	18	1	0	100	

Table 2.5. Area and tenure of forest with at least 5 percent mature eucalypt crown cover and an understorey of secondary species.

	E1	E2	E3	E3+	E3-	E4	E5	TOTAL	%
	ha	ha	ha	ha	ha	ha	ha	ha	
TWWHA	998	5188	1631	1608	3010	1740	0	14175	17
Other State Reserves (National Parks)	3	245	130	133	167	107	0	785	1
Forest Reserves	32	43	29	24	9	3	0	140	<1
Crown land in Conservation Areas	0	198	0	161	207	70	0	636	1
Protected Areas	7	181	19	0	30	53	238	528	1
State Forests	6753	28713	7545	6500	2538	2271	63	54383	67
Unallocated Crown Land	88	1160	214	795	635	398	394	3684	5
HEC-vested land	0	4	0	10	64	1	0	79	
Other Crown Reserves	0	40	0	25	25	15	0	105	<1
Private property	211	1312	27	2911	1572	569	145	6747	8
TOTAL	8092	37084	9595	12167	8257	5227	840	81262	100
PERCENT	10	46	12	15	10	6	1	100	

Discussion

Fig. 2.2 shows that mixed forest occurs primarily in western Tasmania with some outliers in the north-eastern highlands and the Forestier and Tasman Peninsulas. Other small areas of mixed forest are known in eastern Tasmania, e.g. in the Douglas-Apsley National Park, but are not shown in Fig. 2.2 as their PI types indicate T rather than M in the understorey. The distribution of mixed forest is similar to that of rainforest (see Kirkpatrick and Dickinson 1984) but is peripheral to the main blocks of rainforest. Much of the mixed forest occurs in a broad band which trends from north-west to south-east. This band presumably separates a wetter western region where forest wildfires are relatively infrequent from a drier eastern region where forest fires are more common.

Tables 2.2 and 2.3 indicate that almost half the area of mixed forest with a mature myrtle understorey has a eucalypt density of only 5-20 percent. Although it is possible that these stands always had a sparse density of eucalypts, it is much more likely that the eucalypts have become sparse with increasing age as described by Gilbert (1959). This implies that nearly half the mature mixed forest in Tasmania is in the last successional stage prior to becoming rainforest. Although the rate of decrease of eucalypt canopy cover is unknown it is likely that, in the absence of major disturbance, large areas of mixed forest will become rainforest within the next century.

It is noteworthy that 47,000 ha of rainforest has a very sparse eucalypt canopy cover. Eucalypts do not normally regenerate in undisturbed rainforest (Gilbert 1959, Cremer 1960) so it can be assumed that these forests were burnt by a wildfire within the last 400 years. This represents at least 6 percent of the total area of rainforest and is additional to the 8 percent of rainforest which has been burnt since 1950 (Kirkpatrick and Dickinson 1984). Some extensive areas of rainforest regrowth, e.g. in the north-eastern highlands, also provide evidence that a substantial portion of Tasmania's rainforest has been subject to fire.

These results are consistent with the findings of Gilbert (1959) and Jackson (1968) that infrequent wildfire is an integral requirement for the formation of lowland mixed forest. The next chapter compares the floristics of oldgrowth mixed forest with 20-30-year old regrowth which has resulted from either clearfelling, slash burning and sowing with eucalypt seed or from natural regeneration following a wildfire.

3. A COMPARISON OF FLORISTICS OF OLDGROWTH MIXED FORESTS WITH REGENERATION RESULTING FROM LOGGING OR WILDFIRE

Silvicultural regeneration has been established following clearfelling and burning of wet eucalypt forests in Tasmania since the early 1960s. About 50,000 ha have been established on sites with a potential eucalypt height of at least 41m (Wells 1991). Much of this regeneration has been established on sites previously occupied by mixed forest, i.e. oldgrowth eucalypt forest with a rainforest understorey. Silvicultural practices of logging and burning for regeneration cause local increases in the area of wet sclerophyll understoreys at the expense of rainforest understoreys (Cunningham and Cremer 1965). Despite this, some regeneration of rainforest tree species has been reported by Felton and Lockett (1983), Neyland and Hickey (1990), Jordan *et al.* (1992), Tanjung (1992) and in several unpublished studies of regeneration of rainforest tree species after logging of mixed forest in Tasmania which are reviewed in Appendix 2 of this thesis. The review summarises reports by Blakesley (1978), McCormick (1982), Neyland (1983), Heathcote (1985), and unpublished data of Calais which was reported by Hickey and Savva (1992). Information on the regeneration of non-tree species after logging of mixed forest includes the studies of Cunningham and Cremer (1965), Cremer and Mount (1965), Taplin (1982), Taplin *et al.* (1992) and Tanjung (1992) in Tasmania, and of Cook and Drinnan (1984) and Ough and Ross (1992) in Victoria. Information on the early regeneration of unlogged mixed forest burnt by wildfires includes studies by Gilbert (1959), Hill and Read (1984), Ellis (1985) and Barker (1991). Some recent studies of the regeneration of vascular plants following clearfelling and burning of other temperate wet forest types include those of Dickinson and Kirkpatrick (1987) for wet sclerophyll forests in Tasmania, Mueck and Peacock (1992) for wet sclerophyll forest in Victoria and of Halpern (1988, 1989) for *Pseudotsuga* forests in Oregon, USA.

Cunningham and Cremer (1965) examined regeneration survey data from six mixed forest coupes and four wet sclerophyll coupes in southern Tasmania. They presented information on the densities of three rainforest tree species and five wet sclerophyll tree and tall shrub species at the sites both before and 1-3 years after burning. Although they recorded some regeneration of rainforest trees at each of the rainforest coupes they concluded that the post-logging understorey was likely to be dominated by wet sclerophyll scrub for the period of the planned rotation of 80-100 years.

Cremer and Mount (1965) studied the early regeneration of fenced and unfenced sites from 0.5 to 10 years after the clearfelling and burning of both mixed and wet sclerophyll forests in the Florentine Valley. They found that woody species established very soon after the fire and began to dominate the sites 2-10 years after burning. The most abundant species were wet sclerophyll species while the previously common rainforest trees, *Nothofagus cunninghamii* and *Atherosperma moschatum*, were relatively insignificant. They observed that *Atherosperma* seemed to need shade to establish but otherwise there was no succession among woody species in the sense that one stage prepares the site for the next over the period of their study. This observation was described by Noble and Slatyer (1981) as consistent with the initial floristic composition model of succession developed by Egler (1954). The model requires that all species are initially present and there is an apparent succession due to differences in growth and abundance with time.

Taplin (1982) and Taplin *et al.* (1992) described a survey of the occurrence of vascular plants in six silviculturally regenerated coupes in Geeveston District which were between 1 and 17 years old. They reported that the regeneration practices of clearfelling and burning tend to eliminate temperate rainforest tree species, but their conclusions should be treated with caution as none of their surveyed coupes were mixed forest or rainforest prior to being clearfelled and all had been subject to at least one wildfire, and some as many as four wildfires, in the previous 100 years.

Tanjung (1992) examined the stocking of eucalypt, rainforest and sclerophyllous understorey species in mixed forest which was either unlogged, selectively logged and unburnt, or clearfelled, burnt and sown in southwestern Tasmania. The logged and burnt sites included areas burnt two, five, seven and nine years previously. Quadrats within the burnt sites were classed according to the inferred intensity of the regeneration burn. Although some rainforest regeneration occurred on most burnt sites, the stocking rate of rainforest species decreased with increasing fire intensity. The similarity between species present in silvicultural regeneration on clearfelled and burnt sites compared to oldgrowth mixed forest was least, about 50 percent, for sites two years after the fire.

Cook and Drinnan (1984) compared the floristics of silvicultural regeneration aged 4, 10 and 14 years which resulted from the clearfelling of mixed forest dominated by *Eucalyptus nitens* in East Gippsland, Victoria. They reported that the density and order of relative abundance of the common climax species were similar in 14-year-old silvicultural regeneration to that in mature forest. However common epiphytic ferns in the mature forest were rare or absent in the regenerated forests.

Ough and Ross (1992) studied the floristics of wet forest, primarily dominated by *Eucalyptus regnans*, in the Central Highlands of Victoria. The sampled stands ranged in age from silvicultural regeneration less than 21 years old through to oldgrowth forests more than 250 years old. About one third of stands were of mixed age. They concluded that clearfelling resulted in even-aged forest and had a negative effect on the abundance of some species, including at least six species which are prominent in mixed forest in Tasmania. *Grammitis billardierei, Microsorum diversifolium, Nothofagus cunninghamii, Pittosporum bicolor* and *Polystichum proliferum* were rarely recorded in silvicultural regeneration although common in oldgrowth forest. They also noted a decrease in *Dicksonia antarctica* in harvested sites, which was of particular concern as it is an important substrate for a number of epiphytic species. When *Dicksonia* was present in silvicultural regeneration it usually occurred as young sporelings rather than as mature stems.

The composition and/or frequency of vascular plant species in silvicultural regeneration less than 30 years old would be expected to be different from that of oldgrowth mixed forest that is up to 400 years old. A more interesting comparison is between silvicultural regeneration and similar-aged regeneration that has resulted from a wildfire. The latter type would be expected to eventually redevelop, in the absence of further disturbance, into oldgrowth mixed forest according to the successional process postulated by Gilbert (1959). The magnitude of the difference in floristics between the silvicultural and wildfire regeneration will give some indication of the extent to which the normal successional process has been modified by the silvicultural practices of clearfelling, burning and sowing. Such information is needed for both ecological and commercial reasons. It is important ecologically because if regeneration of rainforest species occurs only infrequently it would indicate that silvicultural practices are significantly reducing the extent of current and potential mixed forest. Alternatively, if regeneration of rainforest species is common it would indicate that many wet sclerophyll species and rainforest species are not seriously disadvantaged by silvicultural practices and, in addition to reserved areas, may be conserved in wet eucalypt forests that are logged and regenerated.

The extent of rainforest tree species regeneration is important commercially as mixed forests are a major source of special timbers such as blackwood, myrtle, leatherwood, celery-top pine and sassafras, as well as providing commercial non-timber products such as nectar for leatherwood honey. It is possible that some non-rainforest special timbers species, such as blackwood and silver wattle are abundant in silvicultural regeneration but no collated information on their extent has been presented apart from the study by Jordan *et al.* (1992) of stocking rates in 5-8 year-old coupes.

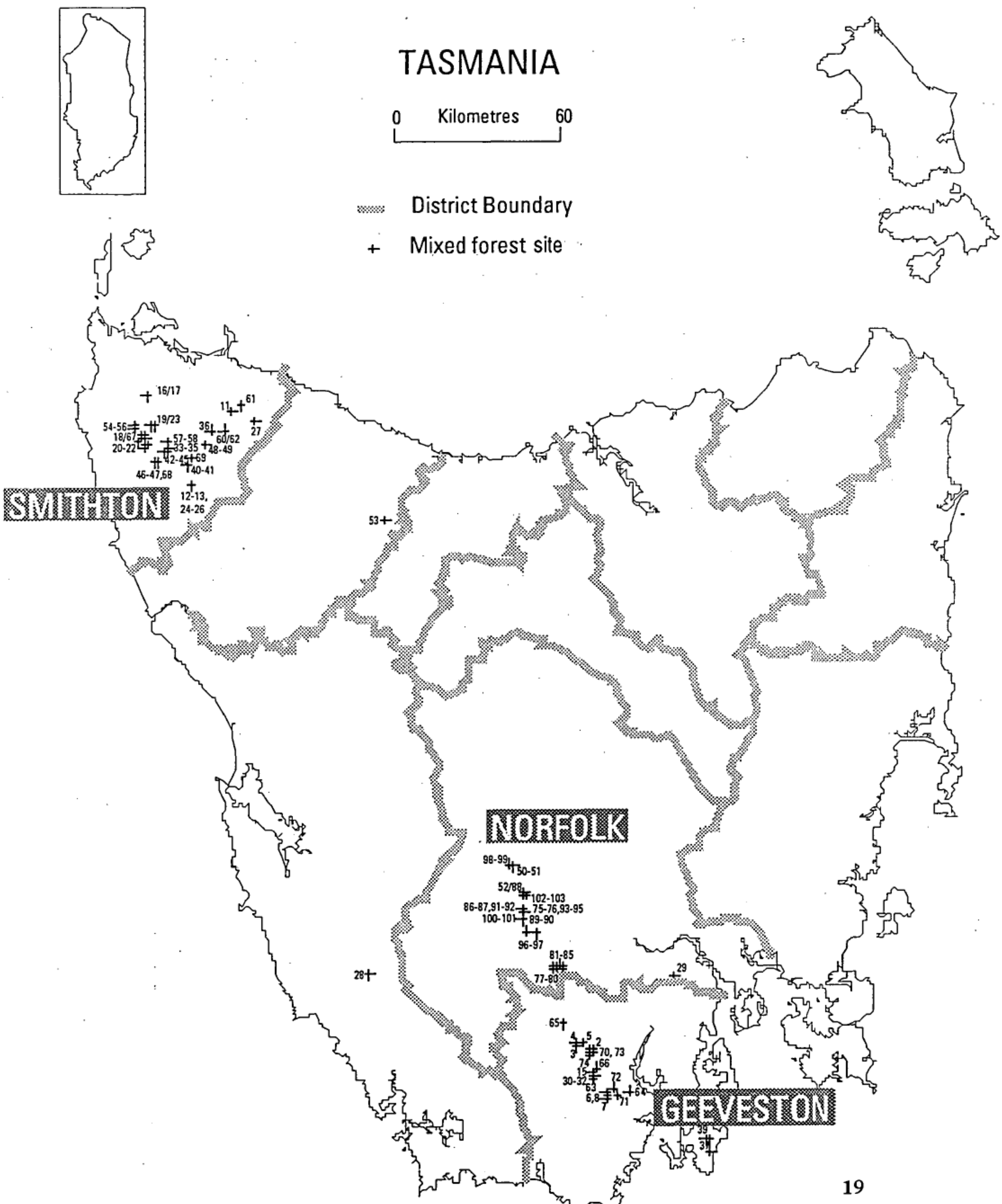
Options for long rotation (e.g. 200 years) management of some wet eucalypt forest areas are being considered to enable an ongoing yield of special timbers (Forestry Commission 1990b). Suitable forest areas would need to have at least a sparse stocking of special timbers species at the time of their establishment to become well-developed mixed forest at 200 years. The Forests and Forest Industry Council (1990) recommended that some areas of mixed forest and rainforest be used for ongoing supplies of special timbers. An alternative strategy currently being tested is the early thinning of wet eucalypt regeneration to enable the production of eucalypt sawlogs on about a 50 year rotation (Kerruish and Rawlins 1991). Such a strategy is likely to disadvantage any potential special timbers component. The consideration of either strategy requires information on the frequency and growth of special timbers species within regenerated mixed forest.

Method

Site selection

The study sites were located in forest patches mapped on Forestry Commission PI maps as either current or former oldgrowth mixed forest, i.e. eucalypt forest more than 110 years old over a rainforest understorey, and with a eucalypt height potential of at least 41m. All sites were below 700 m in elevation and were dominated by either *Eucalyptus obliqua*, *E. regnans* or *E. delegatensis*. *E. brookeriana* and *E. nitida* occurred with *E. obliqua* at some sites. The location of the sites is shown in Fig. 3.1.

Fig. 3.1 Location of sites



Logged sites. The logged sites were all regenerated between 1961 and 1973. Older regeneration was not considered, as clearfelling, burning and sowing only commenced as a standard silvicultural treatment in 1961. The selection of suitable logged sites began with an examination of records of 140 permanent plots, known as CFI-Regen series plots (Forestry Commission 1985) which are established randomly by the Forestry Commission in silvicultural regeneration areas. This program does not extend over all State forests. Sites were located in Geeveston and Smithton Forest Districts as they have:

- mixed forests with rainforest understoreys which cover a diverse range of species and communities (in contrast to the mixed forests of eastern Tasmania which have mainly callidendrous sensu Jarman et al. 1991 understoreys);

- substantial areas of silvicultural regeneration on areas previously occupied by oldgrowth mixed forest;

- CFI-Regen plots located within most areas of silvicultural regeneration;

- had wildfires in areas of oldgrowth mixed forest in the period from 1960 to 1975; and

- relatively few rock types. (Jurassic dolerite is common in Geeveston District and produces relatively fertile forest soils while Pre-Cambrian mudstones are widespread in Smithton District and produce soils of lower fertility (W. Neilsen pers. comm.)).

Sites were also located in the Florentine and Styx valleys within the Norfolk Forest District. This area has substantial tracts of logged mixed forest as well as being the general locality for previous studies of mixed forest including those of Gilbert (1958, 1959) and Mount (1964). Permanent plots in silvicultural regeneration have been established by Australian Newsprint Mills Ltd (ANM) within its pulpwood concession area at a rate of one plot per 100 ha (M. White pers. comm.). ANM plots established prior to 1970 were located randomly while those established since then were located systematically. The latter plots were not considered in this study.

Archived forest-type maps were needed to determine the subset of permanent plots which were previously mixed forest because the pre-logging forest type is not recorded on the plot records. Only thirty-six such plots were located. The number of suitable plots was substantially further reduced by the potential stand height and altitude requirements imposed by this study. Some plots were eliminated because they had not been subject to a regeneration burn.

Sixteen of the logged sites were located within permanent plots established in eucalypt regeneration by the Forestry Commission and five sites were located at permanent plots established by ANM. Permanent plots were used as sites for this study wherever possible because:

- they are randomly located; and

- they provide an opportunity for subsequent floristic descriptions to be made and compared with those gathered by this study so that vegetation changes over several decades may be accurately monitored.

The remaining logged sites were located by using current and archived forest-type maps to identify suitable patches of silvicultural regeneration on sites previously occupied by mixed forest which were within about 200 m of a trafficable road. Sites were selected on maps in the office and a bearing and distance was determined from an identifiable take-off point, e.g. a road intersection or creek crossing. A plot point was established in the forest once the pre-determined distance, usually some multiple of 50 m, had been reached. Occasionally the plot point was unsuitable, e.g. because the immediate vicinity had not in fact been logged. In such cases a new potential plot point was sought by continuing for a further multiple of 50 m on the same bearing. This method, while not truly random, was considered sufficiently objective to overcome potential biases of the measurers.

Wildfire sites. The establishment of sites in regeneration which had resulted from a wildfire in unlogged oldgrowth mixed forest first required an investigation of areas which had burnt in the period from 1960 to 1975. The investigation included checks of annual Forestry Commission Fire Reports for the relevant districts, interviews with district staff with knowledge of the wildfire history over the period, examination of forest-type maps and aerial photographs, and examination of CFI-Regen plots. Unfortunately many areas which initially appeared suitable on forest-type maps, i.e. regenerated areas on former oldgrowth mixed forest sites which were marked as originating from either 'W' (for wildfire) or 'N' (for natural regeneration), were not suitable because they had been thoroughly salvage-logged soon after the fire. Salvage logging had even occurred in some areas within the Mt Field National Park following a wildfire in 1966. Relatively few areas could be identified which were not subsequently logged, and these are documented as Appendix 3. Wildfire sites were located in areas burnt in 1961, 1962, 1966 and 1967. Two of the sites coincided with CFI-Regen plots while the remainder were located in a manner similar to that described above for logged non-CFI sites. Because of their rarity, some wildfire sites were established in remote localities and several sites were usually established within the one locality. Ideally, the 31 wildfire sites would have been located in 31 well separated areas burnt by fires at different times to maximise the range of variation in factors such as seed crops, weather conditions and topography. Practical constraints limited the selection of wildfire sites to six temporally separated fires and twelve locations separated by a distance of at least 5 km. The fires burnt a minimum of 1000 ha although several burnt much larger areas.

Wildfire sites were only established where stands contained eucalypt regeneration and stags (dead standing trees) of rainforest species. Some wildfire sites were observed (but not measured) where the wildfire had been of insufficient intensity to allow eucalypt regeneration or to kill the majority of the rainforest trees. A good example of this was noted at Chrisps Road in the Florentine Valley (Grid reference: Tyenna map sheet 629700) where a rainforest understorey dominated by *Atherosperma moschatum* had regenerated as sprouts under a sparse canopy of fire-damaged oldgrowth eucalypt trees.

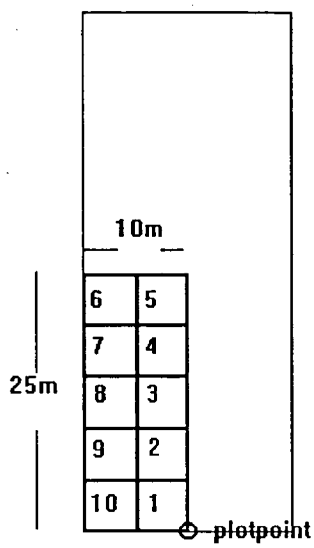
Oldgrowth sites. Oldgrowth mixed forest sites were selected in the manner described for logged non-CFI sites. Suitable areas were usually selected after logged and wildfire sites were established so that the three site types were in the same general locality. Where possible, oldgrowth sites were located in reserves.

Plot layout and measurement.

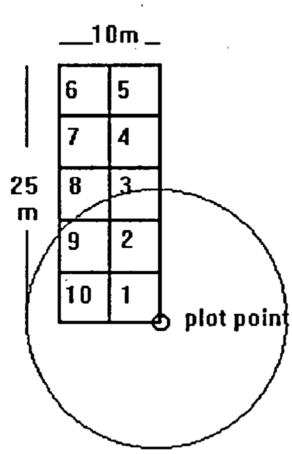
At each site ten contiguous 5 m by 5 m plots were located to form a rectangle 25 m by 10 m (see Fig. 3.2).

Fig. 3.2. Plot layout at (a) a CFI-Regen plot, and (b) an ANM permanent plot.

(a)



(b)



The long axis of the rectangle was located at right angles to the contour of the land after tossing a coin to choose whether to lay the rectangle up or down slope. At sites established on existing CFI-Regen plots the rectangle was located in the nearest left-hand corner of the plot (see Fig. 3.2a). For circular ANM plots the centre-peg was used as the plot-point for this study (Fig. 3.2b).

The maximum slope, aspect, fire intensity and mean dominant height were recorded for each site. The cover of all vascular plants was recorded, using the Braun-Blanquet scale (Mueller-Dombois and Ellenberg 197~~7~~4) on each 5m by 5m plot. Density, diameter and height of the tallest individual were recorded for rainforest tree species, and also for *Acacia melanoxylon*. Only density and diameter were recorded for other trees and tall shrubs. The mode of regeneration, i.e. by seedlings, sprouts or both, was recorded for rainforest trees. A sample site facing sheet and a plot measurement sheet are shown in Appendix 4. Stems of the largest *Phyllocladus aspleniifolius*, *Eucryphia lucida*, *Nothofagus cunninghamii* and *Atherosperma moschatum* were cut from wildfire and logged sites so that their age could be compared with that of the stand. A ring count was also taken from a sclerophyllous woody species, e.g. *Eucalyptus* sp., *Pomaderris apetala* or *Phebalium squameum* at wildfire sites to verify that the stand age at the site was consistent with the recorded age of the wildfire in the locality. One hundred and three sites were established and site information is summarised in Table 3.1. Rock type was determined by electronically overlaying site location data onto the 1: 500,000 Geology Map of Tasmania (Department of Mines 1976).

Table 3.1. Summary information from 103 mixed forest sites. L = logged (silvicultural regeneration), W = wildfire regeneration, and O = oldgrowth mixed forest. *E. bro.* = *Eucalyptus brookeriana, E. del.* = *E. delegatensis, E. nit.* = *E. nitida, E. obl.* = *E. obliqua, E. reg.* = *E. regnans, E. vim.* = *E. viminalis.* Cb = Cambrian basic volcanics, Cm = Cambrian greywackes, Jd = Jurassic dolerite, Ll = Precambrian dolomite, Ls = Precambrian sediments (relatively unmetamorphosed), Ol = Ordovician limestone, Pga = Permian marine sediments, Ptc and Ptf = Triassic non-marine sediments, Ptu and Pu = Parmeener super group, Qh = Quaternary sands and gravels, Tb = Tertiary basalt. The sites are numbered in the order in which field sampling occurred.

Site	Site type	Current PI type	Former PI type	Altitude (m)	Dominant eucalypts	Rock type	CFI/ANM plot no.
1	L	E(66)N/1	E1BMMS	250	E.reg., E.obl.	Jd	1227
2	L	E(67)A/1	E1CMM	165	E.reg.	Jd	1230
3	O	E2dSM3		50	E.obl.	Pu	
4	O	E1dSM3		250	E.obl.	Jd	
5	L	E(71)N/1	E1CMM	90	E.obl.	Jd	1237
6	L	E(63)A/1	o/mE1BMM	400	E.obl.	Jd	2263
7	L	E(72)A/1	o/mE1BM	400	E.del., E.obl.	Jd	2267
8	O	E1dM3TS		390	E. del.	Jd	
9	L	E(69)A/1	o/mE1MM	315	E.del., E.obl.	Ptf	2262
10	L	E(63)A/1	E1CMM	515	E.del., E.obl.	Jd	2264
11	L	E(66)A/2	E2DSM3T	200	E.obl.	Ls	2806
12	O	E2dMT		265	E.obl.	Ls	
13	W	DdE2dER2b	E2CMT	265	E.obl.	Ls	
14	W	DdE2dER2b	E2CMT	270	E.obl.	Ls	
15	W	E1c(Dd)	E1BMM	360	E.obl.	Ptf	
16	L	E(71)A/2	E2+BM2TS	80	E.obl.	Qh	3233
17	O	E2dM-		70	E.obl.	Qh	
18	L	E(71)A/1	E1BMT	110	E.obl.	Cm	4192
19	O	E1cM-		35	E.obl.	Ll	
20	L	E(73)A/1	E1&2CMT	35	E.bro.	Ll	4191
21	L	E(72)A/2	E2&1BMT	55	E.obl.	Cm	4190
22	O	E1cM+		55	E.obl.	Ll	
23	O	E1cM-		40	E.obl.	Ll	
24	O	E2dMT		235	E.obl.	Ls	
25	O	E2dMT		230	E.obl.	Ls	
26	W	DdE2dER2b	E2CMT	270	E.obl.	Ls	
27	O	E1dM-		205	E.obl.	Tb	
28	O	"E3cM-"		60	E.niti.	Qh	
29	W	SDdE2f	E2CT	670	E.reg.	Jd	
30	W	DdE1CER2d	o/mE2A/MM	560	E.del., E.obl.	Jd	
31	W	DdE1cER2bS	E1BMM	510	E.obl.	Jd	
32	W	DdE1CER2bS	E1BMM	360	E.obl.	Ptf	
33	W	ER2pSDdE3f	E2CMT	145	E.obl.	Cb	
34	W	ER2pSDdE3f	E2CMT	110	E.obl.	Cb	

Site	Site type	Current PI type	Former PI type	Altitude (m)	Dominant eucalypts	Rock type	CFI/ANM plot no.
35	W	ER2pSDdE3f	E2DT	90	E.obl.	Cb	
36	W	ER3bSDdE2d	E2CMT	150	E.obl., E. bro.	Ls	
37	L	E(69)A/3	E3CM3TS	325	E.obl.	Jd	2147
38	W	ER2cS/2	E2DMT	370	E.obl.	Jd	
39	O	E3bTM		345	E.obl.	Jd	
40	W	ER2b/3+	E3BMT	160	E.obl.	Ll	
41	W	ER2b/3+	E3BMT	160	E.obl.	Ll	
42	O	E2dM-		140	E.obl.	Cm	
43	O	E2dM-		160	E.obl.	Cm	
44	O	E2bM-		170	E.obl.	Cm	
45	O	E2bM-		170	E.obl.	Cm	
46	O	E2cM-		165	E.obl.	Cm	
47	O	E2DM+		190	E.obl.	Cm	
48	O	E2cTM		180	E.obl.	Ls	
49	O	E2cTM		185	E.obl.	Ls	
50	O	E2cM3T		400	E.reg.	Ptu	
51	O	E2cM3T		420	E.reg.	Ptu	
52	L	E(61)	E1CMT	560	E.reg.	Pga	
53	L	E(68)AW	C/O M3&2TS	610	E.del., E.vim.	Cm	137
54	L	E(62)A	E2BMT	70	E.obl., E. bro.	Ll	
55	L	E(62)A	E2CMT	65	E.obl.	Ll	
56	L	E(64)A	E2CMT	65	E.obl.	Ll	
57	W	ER2BDdE2F	E1CT	170	E.obl.	Cm	
58	W	ER2BDdE2F	E1CT	175	E.obl.	Cm	
59	L	E(69)A	E1B-MT	135	E.obl.	Cm	
60	L	E(70)A	E2BM3T	155	E.obl.	Ls	
61	L	E(72)A	E2DTM3	150	E.obl.	Ls	
62	L	E(70)A	E2BM3T	170	E.obl.	Ls	
63	O	E1DSM2T(Ddf)		430	E.obl.	Ptf	
64	O	E1cM3T		280	E.reg.	Jd	
65	O	E1dTM3		285	E.obl.	Pu	
66	W	DDE2DER2BS	O/ME1CMM	190	E.obl., E.reg.	Jd	
67	L	E(71)A	E1BMT	130	E.obl.	Cm	
68	O	E2dM+		190	E.obl.	Cm	
69	O	E2cM+		90	E.nit.	Ll	
70	W	E(66)W/1	E1aO/mMM	400	E.obl.	Jd	1228
71	O	E1DM3TS		450	E.obl.	Jd	
72	O	E2CM3		550	E.del.	Jd	
73	W	E(66)W/1	E1DMM	360	E.reg.	Jd	1229
74	W	E(66)W/1	E1DMM	360	E.reg., E.obl.	Jd	
75	W	DdE1DER2CS	E1DMT	615	E.obl.	Ol	
76	W	DdE1dER2cS	MTE1F	605	E.reg., E.obl.	Ol	
77	W	F/DE2DER1(p)S	E2CMT	620	E.del., E.reg.	Pu	
78	W	F/DE2DER1(p)S	E1CMT	595	E.reg., E.del.	Pu	
79	W	F/DE2DER1(p)S	E2CMT	610	E.reg., E.del.	Pu	

Site	Site type	Current PI type	Former PI type	Altitude (m)	Dominant eucalypts	Rock type	CFI/ANM plot no.
80	O	E1DM-		470	E.reg.	Pu	
81	L	E(63)A	E1CMT	505	E.reg.	Pu	
82	O	E1BM-		570	E.reg.	Pu	
83	O	E1BM-		585	E.reg.	Pu	
84	L	E(66)A	E1CMT	520	E.reg.	Pu	
85	L	E(66)A	E1CMT	555	E.reg.	Pu	
86	L	E(61)	E1DMT	525	E.reg., E.del.	Ol	
87	L	E(61)	E1DMT	510	E.reg., E.del.	Ol	
88	L	E(61)	E1CMT	550	E.reg.	Ptu	58808438
89	L	E(61)	E1BMT	570	E.del., E.reg.	Ol	58537470
90	L	E(61)	E1DMT	520	E.del.	Ol	58287509
91	L	E(61)	E1CMT	470	E.reg.	Ol	58408409
92	L	E(61)	E2DM3&2T	440	E.reg.	Ol	58408351
93	W	DdE1dER2cS	E1CMT	610	E.reg., E.obl.	Ol	
94	W	DdE1dER2cS	E1CMT	630	E.obl., E.reg.	Ol	
95	O	E1&2bM2TS		480	E.obl., E.del.	Ol	
96	O	E1&2bM2TS		510	E.obl., E.reg.	Ol	
97	W	ER2/1 (p) S	not available	640	E.del.	Ol	
98	O	E2&1dTM3		390	E.reg.	Ptu	
99	O	E2&1dTM3		420	E.reg.	Ptu	
100	W	DdE1dER2cS	MTE1f	670	E.del.	Ol	
101	W	DdE1dER2cS	MTE1f	690	E.del., E.reg.	Ol	
102	O	E2bM-		645	E.reg.	Ptc	
103	O	E2bM-		625	E.reg.	Ptc	

Table 3.2 indicates the number of logged, wildfire and oldgrowth sites in each altitude, dominant eucalypt and rock fertility class.

Table 3.2. Classification of site types according to (a) altitude classes (high = > 500 m, medium = 251-500 m and low = 0-250 m); (b) dominant eucalypt classes (in mixed eucalypt stands the species with the highest density was considered as the dominant eucalypt); (c) rock fertility classes (high = Cambrian basic volcanics, Jurassic dolerite, Tertiary basalt; medium = Cambrian greywacke, Ordovician limestone, Precambrian dolomite, Precambrian sediments (relatively unmetamorphosed), Triassic non-marine sediments; low = Parmeener super group, Permian marine sediments, Quaternary deposits). Fertility ratings as per W. Neilsen (pers. comm.).

(a) altitude classes

	Logged	Wildfire	Oldgrowth	Total
High	11	13	6	30
Medium	6	9	13	28
Low	16	9	20	45
Total	33	31	39	103

(b) dominant eucalypt classes

	Logged	Wildfire	Oldgrowth	Total
E. delegatensis	6	5	2	13
E. obliqua	15	19	25	59
E. regnans	11	7	10	28
other	1		2	3
Total	33	31	39	103

(c) rock fertility classes

	Logged	Wildfire	Oldgrowth	Total
High	7	11	7	25
Medium	20	17	21	58
Low	6	3	11	20
Total	33	31	39	103

Data analysis

The allocations of logged, wildfire and oldgrowth sites according to altitude, dominant eucalypt and rock fertility classes were examined using the Chi-squared distribution to test for significant differences between the numbers of sites in each class.

The site data were entered into an ecological data-handling package called ECOPAK (Minchin 1986) and subsequently manipulated using the updated package called DECODA (Minchin 1990). Three sites, 28, 29 and 53, were deleted from the analysis as they were outliers to the main study areas (see Fig. 3.1). Site 53 was also deemed unsuitable as it was rainforest cut-over for sawlogs prior to being burnt by a wildfire and subsequently sown with eucalypt seed. Therefore it was intermediate in character between a logged and wildfire site. Site variables entered on the database were site type, stand age (for regenerated sites), altitude, aspect, slope, site fertility class, mean annual rainfall, mean rainfall of the driest month, and two variables derived from a principal components analysis of 16 climate variables generated from ESOCLIM. ESOCLIM (Hutchinson 1989) is a computer program that generates monthly and annual estimates of basic climatic parameters for sites specified by spatial reference and elevation. Sixteen climatic parameters were derived for this study. The chosen parameters were similar to those used by BIOCLIM (Busby 1986[a]) to predict species occurrences based on climate. The parameters were derived from ESOCLIM because its climatic surfaces are more up-to-date. A principal components analysis was carried out to reduce the 16 parameters to two independent variables which combined linear combinations of the parameters which best described the variation in the climatic data. Table 3.3 summarises the site variables used for subsequent analyses of floristic trends. A correlation analysis was undertaken, using Spearman's rank correlation coefficient, to examine the relationship between the variables. Species variables were rainforest species status as defined by Jarman *et al.* (1991), and life-form, e.g. tree, tall shrub. The life-form class referred to the predominant habit of the mature plant. For example, an *Atherosperma moschatum* seedling attached to a *Dicksonia antarctica* stem was classed as a tree rather than as an epiphyte.

The mean number of vascular species per site type, within life-form groups, was calculated to compare species richness between oldgrowth and regenerated mixed forest. The numbers of herb, sedge and grass species were excluded from this analysis because several taxa were not identified to species level. An analysis of variance was employed to compare mean species richness for site types. A multiple-range test, based on least significant differences, was used to identify differences between means at the 95 percent confidence level.

Table 3.3. Site variables and descriptions for mixed forest sites.

Variable	Description
SITE TYPE	1 = silvicultural regeneration, 2 = wildfire regeneration and 3 = oldgrowth mixed forest.
AGE	Age of regeneration (site type 1 and 2 only).
ALTITUDE	Altitude in metres from a 1: 25 000 map.
ASPECT	Coded as 1 = S, 2 = E or W, 3 = N.
SLOPE	Maximum slope on site in degrees.
FERTILITY	1 = low fertility (Parmeener super group, Permian marine sediments, Quaternary deposits); 2 = medium fertility (Cambrian greywacke, Ordovician limestone, Precambrian dolomite, Precambrian sediments (relatively unmetamorphosed), Triassic non-marine sediments); 3 = high fertility (Cambrian basic volcanics, Jurassic dolerite, Tertiary basalt). Rock types were determined by reference to the 1:500,000 Geology Map of Tasmania (Mines Department 1976). Fertility ratings as per W. Neilsen (pers. comm.).
ANRAIN	annual rainfall (derived from ESOCLIM)
DRYMON	rainfall of the driest month (derived from ESOCLIM)
CLIMATEPC1	Principal component 1 (derived from a principal components analysis of 16 climatic variables derived from ESOCLIM).
CLIMATEPC2	Principal component 2 (derived from a principal components analysis of 16 climatic variables derived from ESOCLIM).

Mean site occurrences were calculated, e.g. *Nothofagus cunninghamii* occurred at 20 of the 32 logged sites and therefore had a mean site occurrence in silvicultural regeneration of 63 percent. Tests for significant differences between mean site occurrences for the three most common species were undertaken.

Plot data were pooled for each site to determine the relative frequency of each species. Mean frequencies were calculated for each species for each of the three site types and compared graphically using histograms. One-way analyses of variance (ANOVA) were carried out to compare mean frequency estimates for each site type and species. Species which had mean frequencies of less than 5 percent in all of the site types were excluded from the analyses. The comparison of means was based on transformed data in order to stabilise the variances between sites. Prior to the transformation the percentage frequencies were altered so that zero values were represented as 1 and values of 100 were converted to 99 to prevent undefined transformed values. The Logit transformation was used such that:

TFreq=Log{Freq/(100-Freq)}   where TFreq is the transformed frequency score.

When the ANOVAs showed that the null hypothesis (i.e. that the mean frequencies between site types were the same) should be rejected, a multiple-range test was used to indicate which means were significantly different at the 95 percent confidence level.

Percentage cover values were calculated by transforming the Braun-Blanquet classes to their mid-point values as follows: 1=0.5%, 2=3%, 3=15%, 4=37.5%, 5=62.5% and 6=87.5%. The percentage cover values for each plot were pooled to calculate a mean cover value for each species at each site. These data were used for ordination analyses.

Floristic classification and ordination

Multi-variate analyses were carried out to classify and ordinate the sites according to their floristics. Classification was carried out using TWINSPAN (Hill 1979), which is a computer-based divisive, hierarchical clustering technique, and based on the presence or absence of species at the sites. Ordination was carried out using the global form of local non-metric multi-dimensional scaling (LNMDS) which has been shown (Minchin 1987) to be a robust technique for indirect analysis of ecological gradients. The ordination of the 100 sites was carried out by comparing the mean cover abundance for each species at each site. Relationships between the ordination patterns and site variables (see Table 3.3) were examined by vector fitting (*sensu* Kantvilas and Minchin 1989). This technique allows the identification of vectors of maximum correlation for variables within a particular ordination pattern. The LNMDS and vector fitting analyses were performed using DECODA (Minchin 1990).

Results

Chi-squared analyses of the contingency tables shown in Table 3.2 indicated there was no statistically significant association in the allocation of sites between site types or across altitude, dominant eucalypt or rock fertility classes. Thus, the classification of sites was shown to be reasonably balanced.

The occurrence, mean frequency, and mean cover of vascular species on sites in silvicultural regeneration, wildfire regeneration and extant oldgrowth mixed forest are shown in Table 3.4. The species are divided into seven classes according to their life-form. The status of the species as either rainforest, doubtful rainforest or non rainforest species is also shown and follows the classification of Jarman *et al.* (1991).

Table 3.4. Occurrence, mean frequency and mean cover of vascular species in silvicultural regeneration (32 sites), wildfire regeneration (30 sites) and extant oldgrowth mixed forest (38 sites). 1=rainforest species, 2=doubtful rainforest species, 3=non rainforest species. * = non-native species.

Species	silvicultural regeneration			wildfire regeneration			oldgrowth			Rf status
	occ.	mean freq.	mean cover	occ.	mean freq.	mean cover	occ.	mean freq.	mean cover	
Trees	%	%	%	%	%	%	%	%	%	
Acacia dealbata	41	19	1.5	33	11	0.4	3	0	0.0	2
Acacia melanoxylon	44	23	2.5	40	9	1.5	18	4	0.8	2
Acacia mucronata	16	7	1.1	13	9	1.1	-	-	-	2
Acacia verticillata	16	4	0.4	23	9	0.6	-	-	-	2
Acacia riceana	13	6	1.0	-	-	-	-	-	-	3
Atherosperma moschatum	44	10	0.7	43	18	0.6	84	57	9.3	1
Eucalyptus brookeriana	6	5	0.6	3	3	0.4	-	-	-	3
Eucalyptus delegatensis	19	13	2.8	27	20	2.0	5	3	0.6	3
Eucalyptus nitida	-	-	-	3	3	0.0	3	1	0.0	3
Eucalyptus obliqua	63	45	8.7	70	53	10.2	53	23	6.8	3
Eucalyptus ovata	3	3	0.8	-	-	-	-	-	-	3
Eucalyptus regnans	41	30	4.5	37	24	2.6	24	14	2.2	3
Eucryphia lucida	19	4	0.3	40	13	0.9	47	26	4.2	1
Leptospermum lanigerum	9	4	0.5	-	-	-	-	-	-	2
Leptospermum scoparium	25	15	3.1	13	2	0.1	3	0	0.0	2
Melaleuca squarrosa	19	7	0.6	7	5	0.5	3	1	0.1	2
Nothofagus cunninghamii	63	30	3.7	80	41	2.2	95	76	15.4	1
Phyllocladus aspleniifolius	38	16	0.8	40	21	0.9	45	11	0.8	1
Tall shrubs										
Anodopetalum biglandulosum	13	5	0.3	7	1	0.0	26	24	6.4	1
Anopterus glandulosus	9	6	0.7	17	4	0.3	21	13	1.5	1
Banksia marginata	3	0	0.0	3	0	0.0	-	-	-	3
Bedfordia salicina	3	1	0.1	3	2	0.1	-	-	-	3
Cassinia aculeata	13	4	0.1	10	2	0.0	-	-	-	3
Cenarrhenes nitida	13	4	0.1	23	7	0.2	32	10	0.9	1
Hakea lissosperma	-	-	-	3	1	0.0	-	-	-	3
Monotoca glauca	63	30	2.2	70	31	2.4	16	3	0.0	1
Notelaea ligustrina	6	1	0.0	7	3	0.1	5	2	0.1	1
Olearia argophylla	34	20	4.7	47	30	3.2	26	10	1.5	1
Phebalium squameum	41	23	2.4	57	31	4.7	5	2	0.2	2
Pittosporum bicolor	22	5	0.2	23	6	0.3	53	11	0.5	1
Pomaderris apetala	41	32	8.9	63	47	10.7	13	5	0.4	2
Prostanthera lasianthos	3	0	0.0	17	5	0.7	-	-	-	2
Tasmannia lanceolata	25	9	0.5	33	13	0.7	29	9	0.3	1
Zieria arborescens	9	4	0.4	10	7	0.7	5	1	0.0	2

Species	silvicultural regeneration			wildfire regeneration			oldgrowth			Rf status
	occ.	mean freq.	mean cover	occ.	mean freq.	mean cover	occ.	mean freq.	mean cover	
Low shrubs	%	%	%	%	%	%	%	%	%	
Aristotelia peduncularis	16	5	0.0	33	9	0.1	26	6	0.0	1
Bauera rubioides	9	3	0.5	3	2	0.0	-	-	-	2
Coprosma hirtella	3	1	0.0	-	-	-	-	-	-	3
Coprosma nitida	3	0	0.0	-	-	-	8	1	0.0	1
Coprosma quadrifida	44	15	0.3	17	7	0.1	26	8	0.1	1
Correa lawrenciana	3	0	0.0	-	-	-	3	2	0.2	2
Cyathodes glauca	19	12	0.6	23	12	0.5	13	5	0.4	2
Cyathodes juniperina	28	11	0.3	43	14	0.2	18	4	0.2	1
Cyathodes parvifolia	-	-	-	3	0	0.0	-	-	-	1
Gaultheria hispida	3	1	0.0	20	3	0.0	3	0	0.0	1
Helichrysum backhousii	3	1	0.1	-	-	-	-	-	-	3
Helichrysum dendroideum	3	1	0.1	-	-	-	-	-	-	3
Olearia lirata	16	4	0.2	3	2	0.0	-	-	-	3
Olearia persoonioides	6	1	0.1	-	-	-	5	1	0.0	1
Olearia stellulata	3	0	0.0	-	-	-	-	-	-	3
Orites diversifolia	6	2	0.1	3	1	0.0	5	2	0.1	1
Oxylobium ellipticum	-	-	0.0	3	3	-	-	-	-	3
Persoonia juniperina	3	1	0.0	-	-	-	-	-	-	3
Pimelea cinerea	6	1	0.0	20	7	0.1	3	0	0.0	1
Pimelea drupacea	63	19	0.1	70	32	0.2	50	11	0.1	1
Pseudopanax gunnii	6	1	0.0	-	-	-	-	-	-	1
Richea dracophylla	3	3	0.2	-	-	-	-	-	-	3
Senecio linearifolius	3	0	0.0	7	1	0.3	3	0	0.6	3
Trochocarpa cunninghamii	3	0	0.1	23	12	-	16	7	0.2	1
Trochocarpa disticha	16	5	0.0	-	-	-	13	5	0.0	1
Trochocarpa gunnii	3	1	0.0	-	-	0.0	3	1	0.0	1
Herbs, sedges and grasses										
Acaena novae-zelandiae	13	1	0.0	3	1	0.0	-	-	-	3
Acianthus viridis	-	-	-	-	-	-	3	0	0.0	1
Carex sp.	6	1	0.0	-	-	-	-	-	-	1
Corybas spp.	-	-	-	-	-	-	3	0	0.0	2
Dianella tasmanica	6	3	0.1	10	7	0.1	3	2	0.0	1
Drymophila cyanocarpa	9	1	0.1	23	3	0.0	5	1	0.0	1
Epilobium spp.	6	1	0.0	-	-	-	-	-	-	3
Gahnia grandis	72	52	6.6	43	24	2.2	24	5	0.1	1
Galium australe	6	1	0.0	13	1	0.0	-	-	-	1
Gnaphalium collinum	3	0	0.0	-	-	-	-	-	-	3
Grass spp.	9	3	0.0	7	2	0.0	-	-	-	3
Hydrocotyle spp.	28	8	0.0	20	13	0.1	8	2	0.0	1
*Hypochoeris radicata**	3	0	0.0	-	-	-	-	-	-	3

Species	silvicultural regeneration			wildfire regeneration			oldgrowth			Rf status
	occ.	mean freq.	mean cover	occ.	mean freq.	mean cover	occ.	mean freq.	mean cover	
	%	%	%	%	%	%	%	%	%	
Juncus spp.	16	4	0.1	-	-	-	-	-	-	3
Lepidosperma elatius	-	-	-	17	6	0.2	11	3	0.1	2
Oxalis corniculata	3	0	0.0	-	-	0.0	-	-	0.0	3
Orchid spp.	3	1	-	7	1	0.2	3	0	-	3
*Rubus fruticosus**	3	0	0.0	-	-	-	-	-	-	3
Uncinia spp.	6	2	0.0	17	3	0.0	5	1	0.0	1
Urtica incisa	3	0	0.0	-	-		-	-		1
Epiphytic ferns										
Asplenium flaccidum	3	0	0.0	-	-	-	16	3	0.0	1
Ctenopteris heterophylla	-	-	-	20	3	0.0	24	4	0.0	1
Grammitis billardierei	28	4	0.0	50	20	0.1	92	59	0.3	1
Hymenophyllum australe	-	-	-	10	2	0.0	18	5	0.0	1
H. cupressiforme	9	1	0.0	30	16	0.1	39	10	0.1	1
H. flabellatum	6	1	0.0	10	6	0.1	47	19	0.2	1
H. marginatum	-	-	-	-	-	-	3	0	0.0	1
H. peltatum	3	0	0.0	17	4	0.0	37	13	0.1	1
H. rarum	22	3	0.0	43	22	0.1	95	56	0.3	1
Microsorum diversifolium	28	5	0.0	53	14	0.1	61	13	0.1	1
Polyphlebium venosum	9	1	0.1	17	2	0.0	16	2	0.1	1
Rumohra adiantiformis	44	11	0.1	53	21	0.3	68	19	0.3	1
Tmesipteris billardierei	-	-	-	13	2	0.0	29	9	0.1	1
Ground ferns										
Blechnum fluviatile	-	-	-	-	-	-	3	0	0.2	1
Blechnum nudum	38	13	0.5	3	0	0.0	5	1	0.2	1
Blechnum wattsii	53	22	0.5	57	28	0.8	84	55	10.0	1
Cyathea australis	-	-	-	-	-	-	3	0	0.0	1
Dicksonia antarctica	81	36	4.2	83	45	4.5	79	49	7.5	1
Gleichenia microphylla	6	2	0.1	3	1	0.0	-	-	-	1
Histiopteris incisa	88	46	1.7	93	44	1.3	50	16	0.4	1
Hypolepis rugosula	56	15	0.5	47	10	0.1	18	4	0.0	1
Polystichum proliferum	66	33	1.2	87	44	1.5	53	27	5.3	1
Pteridium esculentum	81	50	3.9	73	34	1.8	5	1	0.0	2
Sticherus tener	6	3	0.0	3	0	0.0	-	-	-	1
Climbers										
Billardiera longiflora	22	11	0.1	17	6	0.0	3	1	0.0	3
Clematis aristata	34	11	0.2	27	8	0.1	11	2	0.0	1
Prionotes cerinthoides	-	-	-	10	2	0.0	5	5	0.1	1
Sarcochilus australis	-	-	-	3	1	0.0	8	1	0.0	3

Eighteen tree, 16 tall shrub, 26 low shrub, 13 epiphytic fern, 11 ground fern and 4 climber species were recorded. Twenty taxa of herbs, sedges and grasses were recorded although the species number would be greater as several taxa were only identified to genus level. The total number of taxa for all life-forms was 108 but this is reduced to 57 if species with a mean frequency of less than 5 percent in any of the site types are excluded. The occurrence of non-native species was minimal and confined to *Hypochoeris radicata* and *Rubus fruticosus* which each occurred at one site in silvicultural regeneration.

Climate profile of the sites

Table 3.5 provides summary statistics for the 16 climatic parameters generated for the 100 sites. The principal components analysis indicated that the first two components described 66 percent and 22 percent of the variation in the climatic data. A biplot was drawn (see Fig. 3.3) to indicate how each of the original climatic parameters contributes to these principal components. Each parameter is represented by a line whose length and direction is proportional to the contribution of that parameter to the derived principal components. The biplot shows that many of the parameters are highly correlated. In fact, two pairs of variables, MEANHOT (mean temperature of the warmest quarter), MEANDRY (mean temperature of the driest quarter) and DRYQUT (rainfall of the driest quarter), HOTQUT (rainfall of the warmest quarter) are identical for this set of sites. The biplot also shows that the first principal component is most positively influenced by the temperature variables ANRANGE, DRYMON and DRYQUT (=HOTQUT) and most negatively influenced by eight variables including MAXMON, SEASON, MEANDRY, MEANWET, MEANHOT, MINMON, MEANTEMP and MEANCOLD. The second principal component is most influenced by annual rainfall (ANRAIN).

Table 3.5. Mean, minimum and maximum values for climatic parameters for the 100 sites.

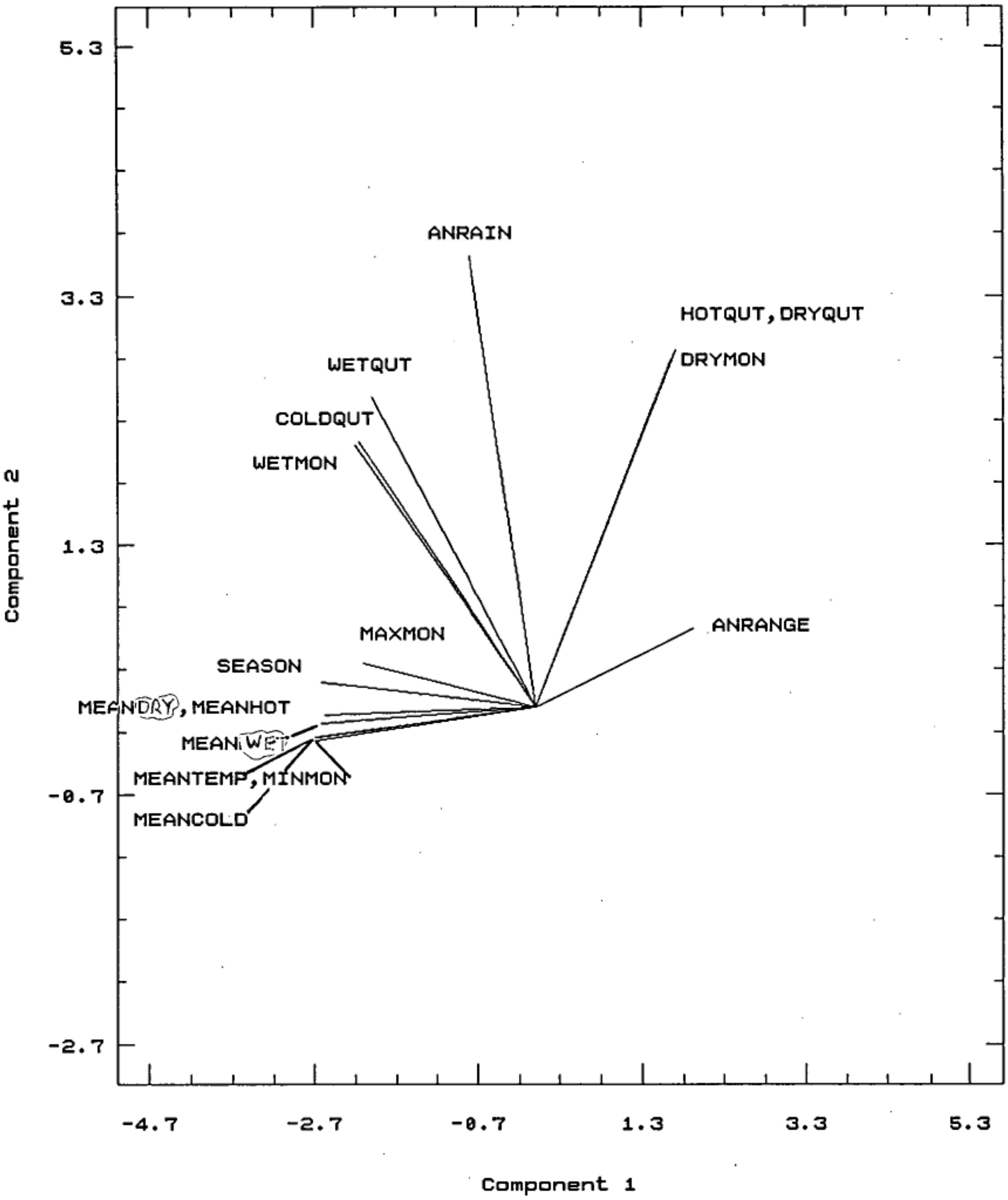
Parameter	Variable name	mean	std dev.	minimum	maximum
annual temperature range (^0C)	ANRANGE	16.8	1.23	14.4	19.8
mean annual temperature (^0C)	MEANTEMP	10.1	1.41	7.8	12.3
annual maximum monthly max. temp. (^0C)[1]	MAXMON	19.9	1.03	17.9	21.8
annual minimum monthly minimum temp.(^0C)	MINMON	3.1	1.69	0.8	5.8
mean temp. of coldest quarter (^0C)[2]	MEANCOLD	6.7	1.70	4.0	9.4
mean temp. of wettest quarter (^0C)	MEANWET	7.0	1.55	4.4	9.3
mean temp. of warmest quarter (^0C)	MEANHOT	14.0	1.24	11.9	15.7
mean temp. of driest quarter (^0C)	MEANDRY	14.0	1.24	11.9	15.7
annual rainfall (mm)	ANRAIN	1431.5	123.69	1145.8	1748.2
rainfall of the wettest month (mm)	WETMON	166.4	27.75	109.97	224.92
rainfall of the driest month (mm)	DRYMON	64.1	7.11	52.9	81.8
seasonality[3]	SEASON	28.1	7.00	13.4	37.0
rainfall of the driest quarter (mm)	DRYQUT	222.9	24.52	187.2	278.2
rainfall of the wettest quarter (mm)	WETQUT	468.9	63.5	320.9	613.3
rainfall of the warmest quarter (mm)	HOTQUT	222.9	24.52	187.2	278.2
rainfall of the coldest quarter (mm)	COLDQUT	451.8	70.99	320.9	613.3

[1] the mean maximum temperature for the hottest month of the year

[2] the quarterly parameters are running quarters of any three consecutive months

[3] the coefficient of variance for temperatures for each month of the year

Fig. 3.3. Biplot of the contribution of 16 climatic parameters to the first two principal components.



Climatic variables which have been traditionally used to indicate suitable sites for mixed forest or rainforest include annual rainfall and rainfall of the driest month. Jackson (1983) suggested that the minimum rainfall required for rainforest in Tasmania appeared to be about 800 mm per year and a summer minimum of no less than 25 mm per month during January, February and March. Under these conditions rainforest is confined to southeastern aspects of dissected gullies. Where annual rainfall is about 1200 mm and the mean monthly summer rainfall is more than 40 mm, rainforest can occur over a wide range although frequently it is replaced by wet sclerophyll elements due to disturbance by fire (Jackson 1983). Ashton (1981[a]) indicated that wet eucalypt forests are common in areas of southeastern Australia which receive between 1000-1500 mm but that rainforest elements were lacking, or rare, for areas receiving only 25-30 mm in the driest month. Busby (1986[b]) estimated the climate profile of *Nothofagus cunninghamii* from 209 sites in Tasmania and 124 sites in Victoria and reported a range in annual rainfall of 930 to 3520 mm and in rainfall of the driest month of 51 to 108 mm. Variable DRYMON in Table 3.5 indicates that all the sites in this study had a minimum monthly rainfall exceeding 50 mm and therefore can be considered inside the range for the development of rainforest elements in wet eucalypt forest. Variable ANRAIN showed that the mean annual rainfall was 1430 mm with a minimum value of about 1150 mm.

Correlations between site variables

The correlation analysis of the site variables is summarised in Table 3.6.* As expected, SITETYPE was not correlated with any other variables. ALTITUDE was strongly correlated with CLIMATEPC1 and weakly correlated with SLOPE, CLIMATEPC2 and FERTILITY. ASPECT showed no significant correlation with other variables. SLOPE was weakly correlated with the climatic variables, CLIMATEPC1, DRYMON and ANRAIN. Significant correlations exist between the climatic variables with the strongest correlation being between ANRAIN and CLIMATEPC2.

* Only significant correlations ($p < 0.05$) are shown. No adjustment was made for the effect of multiple-comparisons.

Table 3.6 Rank (Spearman) correlation matrix for site variables (only significant correlations, with a p value less than 0.05, are shown).

		1	2	3	4	5	6	7	8
1	SITETYPE								
2	ALTITUDE	-							
3	ASPECT	-	-						
4	SLOPE	-	0.51	-					
5	FERTILITY	-	-0.20	-	-				
6	ANRAIN	-	-	-	-0.12	-			
7	DRYMON	-	0.49	-	0.31	0.31	0.44		
8	CLIMATEPC1	-	0.93	-	0.52	-0.52	-0.29	0.60	
9	CLIMATEPC2	-	-	-	-	-	0.94	0.72	-

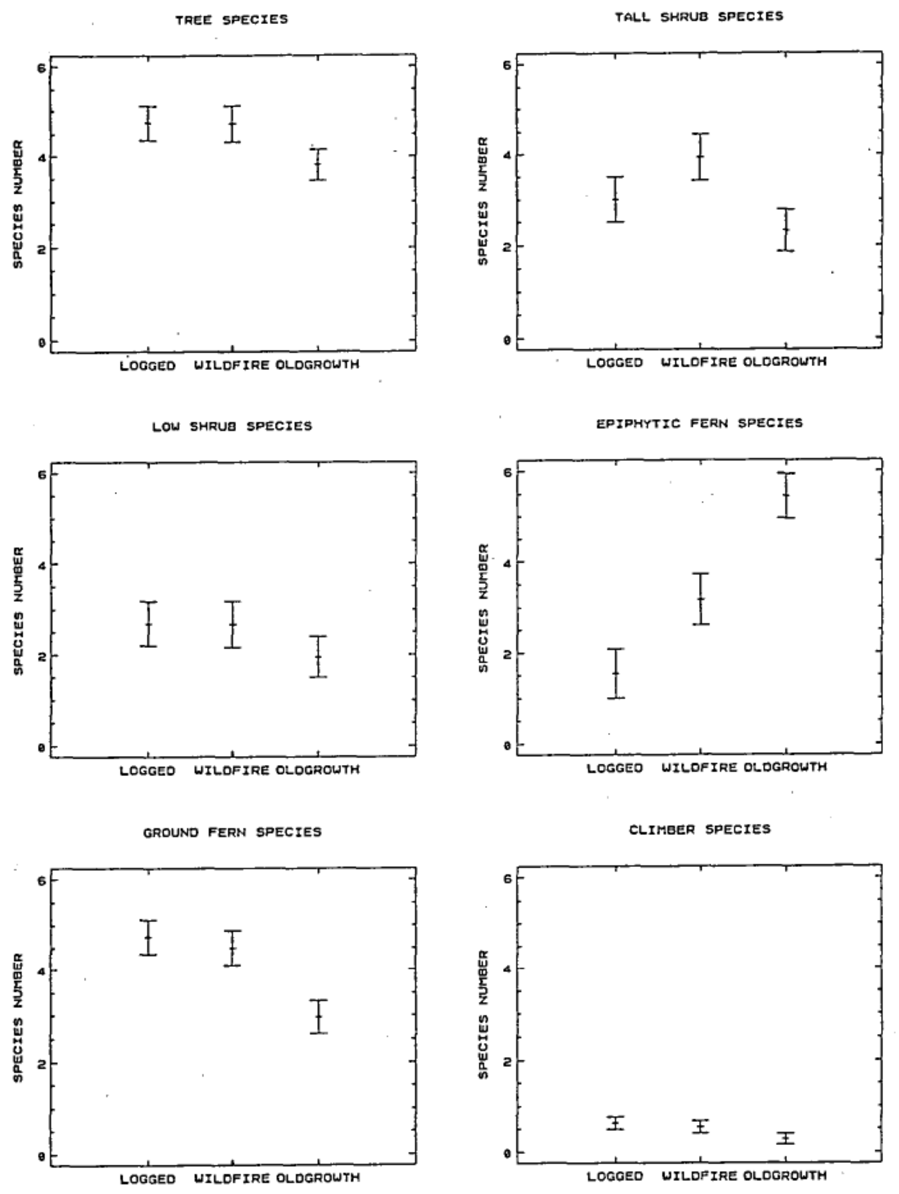
Comparison of species richness

Table 3.7 summarises the comparison of means for life-form groups and shows a similar pattern for species richness of trees, ground ferns and climbers. Each of these life-form groups had similar numbers of species in silvicultural regeneration and wildfire regeneration but significantly lower numbers in oldgrowth mixed forest. The differences in numbers of low shrub species were not statistically significant. The mean number of tall shrub species on logged sites was similar to the number on wildfire and oldgrowth sites but the means for wildfire and oldgrowth sites were significantly different. The greatest difference in site types was apparent in the number of epiphytic fern species where the means for all three site types were significantly different. Silvicultural regeneration had the least diversity of epiphytic ferns while oldgrowth forest was the most diverse. This trend was the reverse of that shown by the other life-forms. The trends are more easily seen in Fig. 3.4 which shows mean species richness, and 95 percent confidence intervals, for life-form groups which have statistically significant differences between site types.

Table 3.7. Life-form group, mean species richness and significance of differences between site types (site types with the same superscript have similar means at the 95% confidence level), p-value for the hypothesis that the means are the same. N.S.=non significant.

life-form	mean species richness			p-value
	logged	wildfire	oldgrowth	(means)
trees	4.8^{A}	4.7^{A}	3.8^{B}	<0.05
tall shrubs	3.0^{AB}	3.9^{B}	2.3^{A}	<0.01
low shrubs	2.7	2.7	1.9	N.S.
epiphytic ferns	1.5^{A}	3.2^{B}	5.5^{C}	<0.001
ground ferns	4.8^{A}	4.7^{A}	3.0^{B}	<0.001
climbers	0.6^{A}	0.5^{A}	0.3^{B}	<0.05

Fig. 3.4. Species richness versus site type (i.e. logged, wildfire or oldgrowth sites).



Comparison of site occurrence for ubiquitous oldgrowth species

The tree species *Nothofagus cunninghamii,* and the epiphytic ferns, *Hymenophyllum rarum* and *Grammitis billardierei,* were the most common oldgrowth species and occurred at more than 90 percent of oldgrowth mixed forest sites. The hypothesis that the proportion of 'successes', i.e. the presence of each of these species, was the same for each forest type was tested and rejected. A subsequent test was applied to the hypothesis that the expected proportions of successes in silvicultural regeneration and wildfire regeneration are the same (see Table 3.8). The 95 percent confidence interval for the expected difference in proportions was also calculated.

Table 3.8. P-values for the hypothesis that the expected proportion of the occurrence of a species is the same for logged (silvicultural regeneration) and wildfire (wildfire regeneration) sites. The 95% confidence interval for the expected difference in proportions is also shown.

Species	logged sites		wildfire sites		p-value	confidence interval
	present	total	present	total		
Nothofagus cunninghamii	20	32	24	30	N.S.	-0.40 to 0.05
Hymenophyllum rarum	7	32	13	30	N.S.	-0.44 to 0.01
Grammitis billardierei	9	32	15	30	N.S.	-0.46 to 0.09

Table 3.8 indicates that the proportion of silvicultural regeneration sites which contained the three most common rainforest species in oldgrowth mixed forest was not significantly different from the proportion of wildfire regeneration sites which contained those species.

Comparison of mean frequency

Tables 3.9-3.15 show the results of the significance tests for each life-form type. Fig. 3.5 shows the untransformed mean frequency, and 95 percent confidence intervals (derived from the within-treatment standard errors) for each species for each life-form type. The species names have been abbreviated and are ordered alphabetically.

Table 3.9. Results of significance tests for differences between transformed mean frequencies of tree species (1 = rainforest species, 2 = doubtful rainforest species, 3 = non rainforest species) for each site type (site types with the same letter have statistically similar means at the 95% confidence level, N.S.=non significant).

Species	Rainforest status	p-value (ANOVA)	Comparison of means by site type logged	wildfire	oldgrowth
Acacia dealbata	2	<0.001	A	A	B
Acacia melanoxylon	2	<0.01	A	B	B
Acacia mucronata	2	N.S.			
Acacia riceana	3	<0.05	A	B	B
Acacia verticillata	2	<0.05	AB	A	B
Atherosperma moschatum	1	<0.001	A	A	B
Eucalyptus brookeriana	3	N.S.			
Eucalyptus delegatensis	3	<0.05	AB	A	B
Eucalyptus obliqua	3	<0.01	A	A	B
Eucalyptus regnans	3	N.S.			
Eucryphia lucida	1	<0.01	A	AB	B
Leptospermum scoparium	2	<0.01	A	B	B
Melaleuca squarrosa	2	N.S.			
Nothofagus cunninghamii	1	<0.001	A	A	B
Phyllocladus aspleniifolius	1	N.S.			

Table 3.10. Results of significance tests for differences between transformed mean frequencies of tall shrub species.

Species	Rainforest status	p-value (ANOVA)	Comparison of means by site type logged	wildfire	oldgrowth
Anodopetalum biglandulosum	1	<0.01	A	A	B
Anopterus glandulosus	1	N.S.			
Cenarrhenes nitida	1	N.S.			
Monotoca glauca	1	<0.001	A	A	B
Olearia argophylla	1	N.S.			
Phebalium squameum	2	<0.001	A	A	B
Pittosporum bicolor	1	<0.05	A	A	B
Pomaderris apetala	2	<0.001	A	A	B
Prostanthera lasianthos	2	<0.01	A	B	A
Tasmannia lanceolata	1	N.S.			
Zieria arborescens	2	N.S.			

Table 3.11. Results of significance tests for differences between transformed mean frequencies of low shrub species (1 = rainforest species, 2 = doubtful rainforest species, 3 = non rainforest species) for each site type (site types with the same letter have statistically similar means at the 95% confidence level, N.S.=non significant).

Species	Rainforest status	p-value (ANOVA)	Comparison of means by site type logged	wildfire	oldgrowth
Aristotelia peduncularis	1	N.S.			
Coprosma quadrifida	1	N.S.			
Cyathodes glauca	2	N.S.			
Cyathodes juniperina	1	N.S.			
Pimelea cinerea	1	<0.05	AB	B	A
Pimelea drupacea	1	<0.05	AB	B	A
Trochocarpa cunninghamii	1	N.S.			
Trochocarpa disticha	1	N.S.			

Table 3.12. Significance tests for differences between transformed mean frequencies of herb and sedge species.

Species	Rainforest status	p-value (ANOVA)	Comparison of means by site type logged	wildfire	oldgrowth
Dianella tasmanica	1	N.S.			
Gahnia grandis	1	<0.001	A	B	C
Hydrocotyle spp.	1	N.S.			
Lepidosperma elatius	2	N.S.			

Table 3.13. Results of significance tests for differences between transformed mean frequencies of epiphytic fern species (1 = rainforest species, 2 = doubtful rainforest species, 3 = non rainforest species) for each site type (site types with the same letter have statistically similar means at the 95% confidence level, N.S.=non significant).

Species	Rainforest status	p-value (ANOVA)	Comparison of means by site type: logged	wildfire	oldgrowth
Grammitis billardierei	1	<0.001	A	B	C
Hymenophyllum australe	1	<0.05	A	AB	B
H. cupressiforme	1	<0.05	A	B	B
H. flabellatum	1	<0.001	A	A	B
H. peltatum	1	<0.001	A	A	B
H. rarum	1	<0.001	A	B	C
Microsorum diversifolium	1	<0.05	A	B	B
Rumohra adiantiformis	1	N.S.			
Tmesipteris billardierei	1	<0.01	A	A	B

Table 3.14. Significance tests for differences between transformed mean frequencies of ground fern species.

Species	Rainforest status	p-value (ANOVA)	Comparison of means by site type: logged	wildfire	oldgrowth
Blechnum nudum	1	<0.001	A	B	B
Blechnum wattsii	1	<0.001	A	A	B
Dicksonia antarctica	1	N.S.			
Histiopteris incisa	1	<0.001	A	A	B
Hypolepis rugosula	1	<0.01	A	A	B
Polystichum proliferum	1	N.S.			
Pteridium esculentum	2	<0.001	A	~~B~~ A	B

Table 3.15. Significance tests for differences between transformed mean frequencies of climber species.

Species	Rainforest status	p-value (ANOVA)	Comparison of means by site type: logged	wildfire	oldgrowth
Billardiera longiflora	3	<0.05	A	AB	B
Clematis aristata	1	<0.05	A	AB	B
Prionotes cerinthoides	1	N.S.			

Fig. 3.5(a). Mean frequency of tree species in silvicultural regeneration (= logged), wildfire regeneration (= wildfire) and oldgrowth mixed forest. (see Appendix 5 for a key to abbreviated species names)

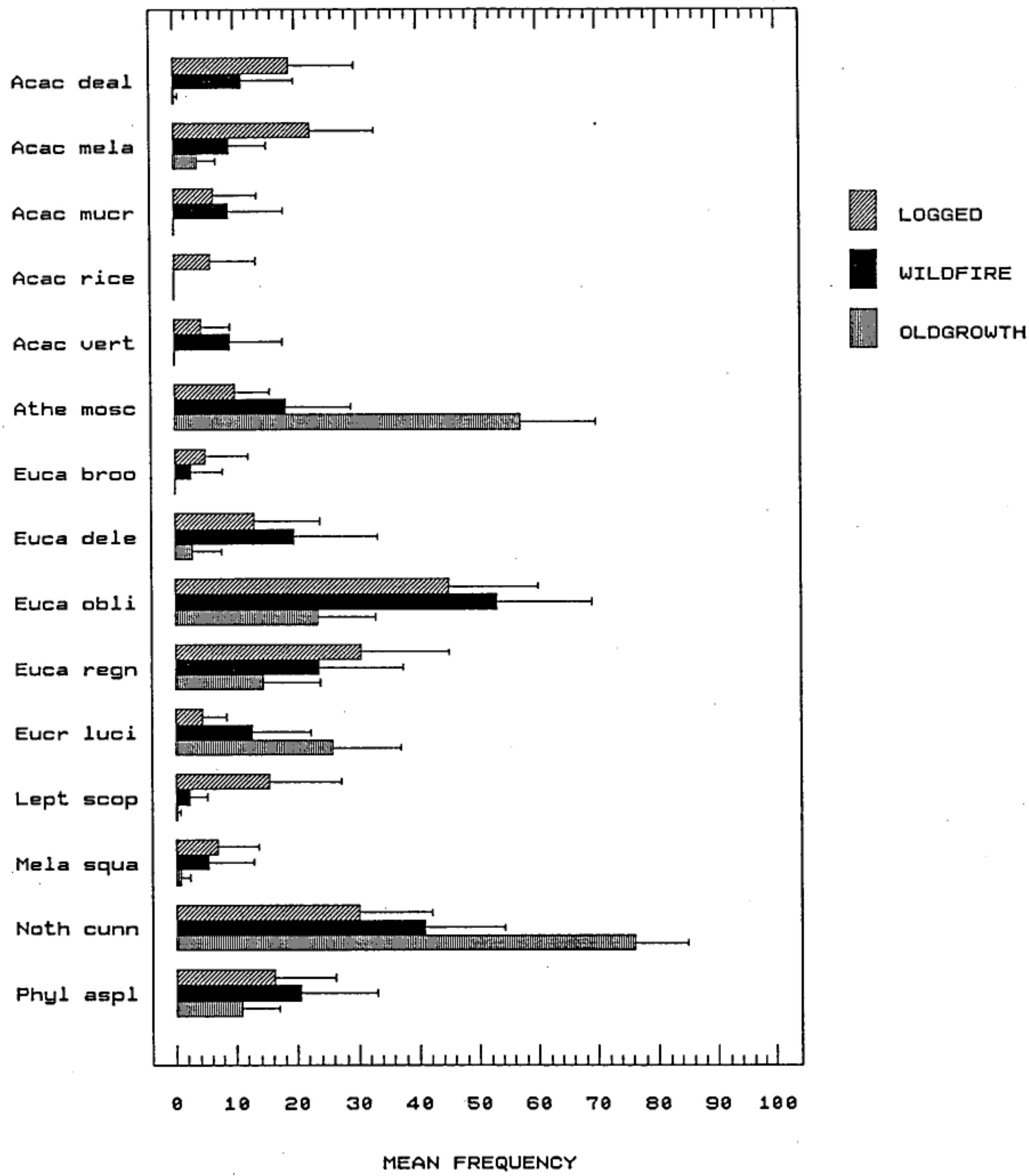


Fig. 3.5(b). Mean frequency of tall shrub species in silvicultural regeneration (= logged), wildfire regeneration (= wildfire) and oldgrowth mixed forest. (see Appendix 5 for a key to abbreviated species names)

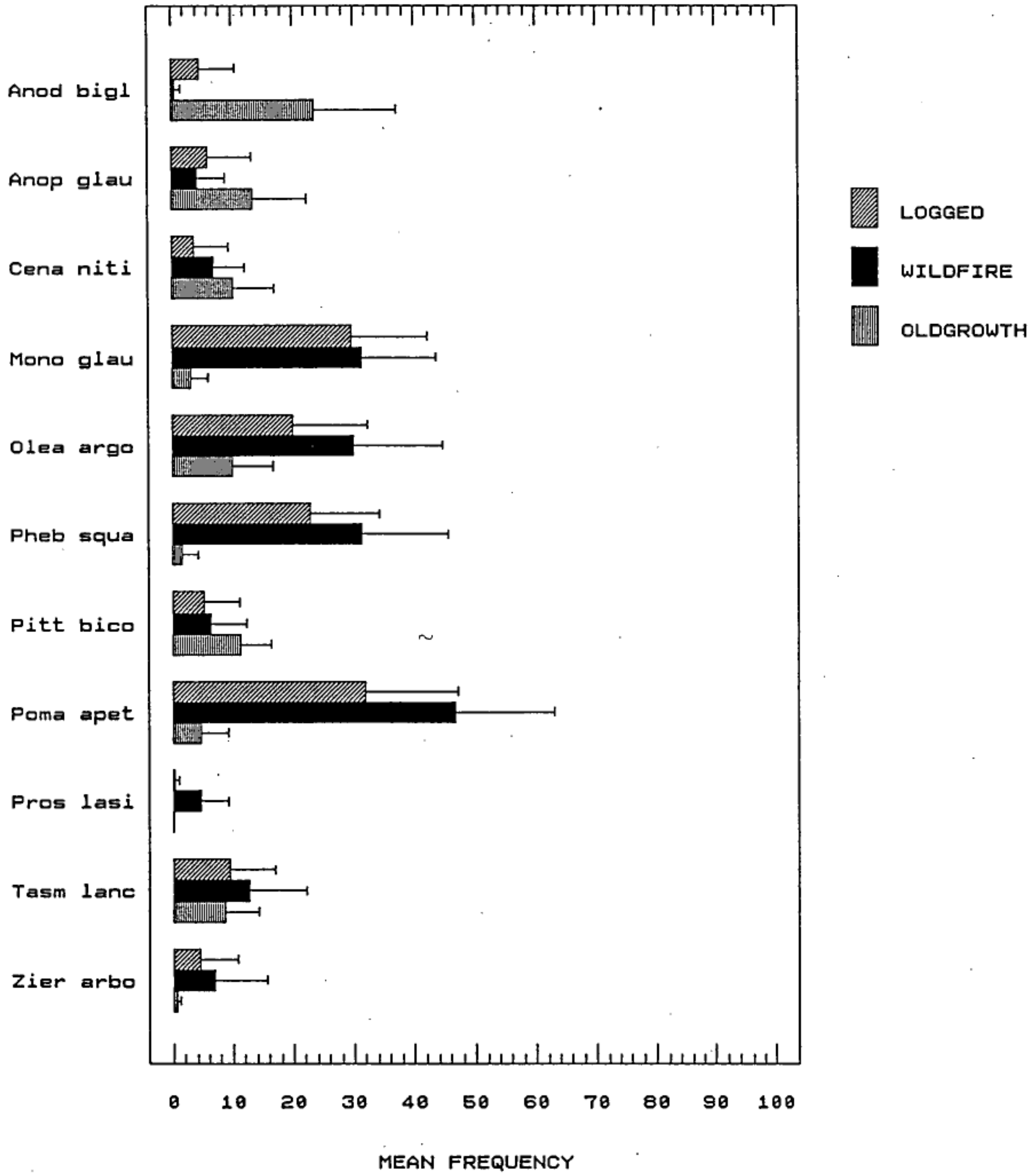


Fig. 3.5(c). Mean frequency of low shrub species in silvicultural regeneration (= logged), wildfire regeneration (= wildfire) and oldgrowth mixed forest. (see Appendix 5 for a key to abbreviated species names)

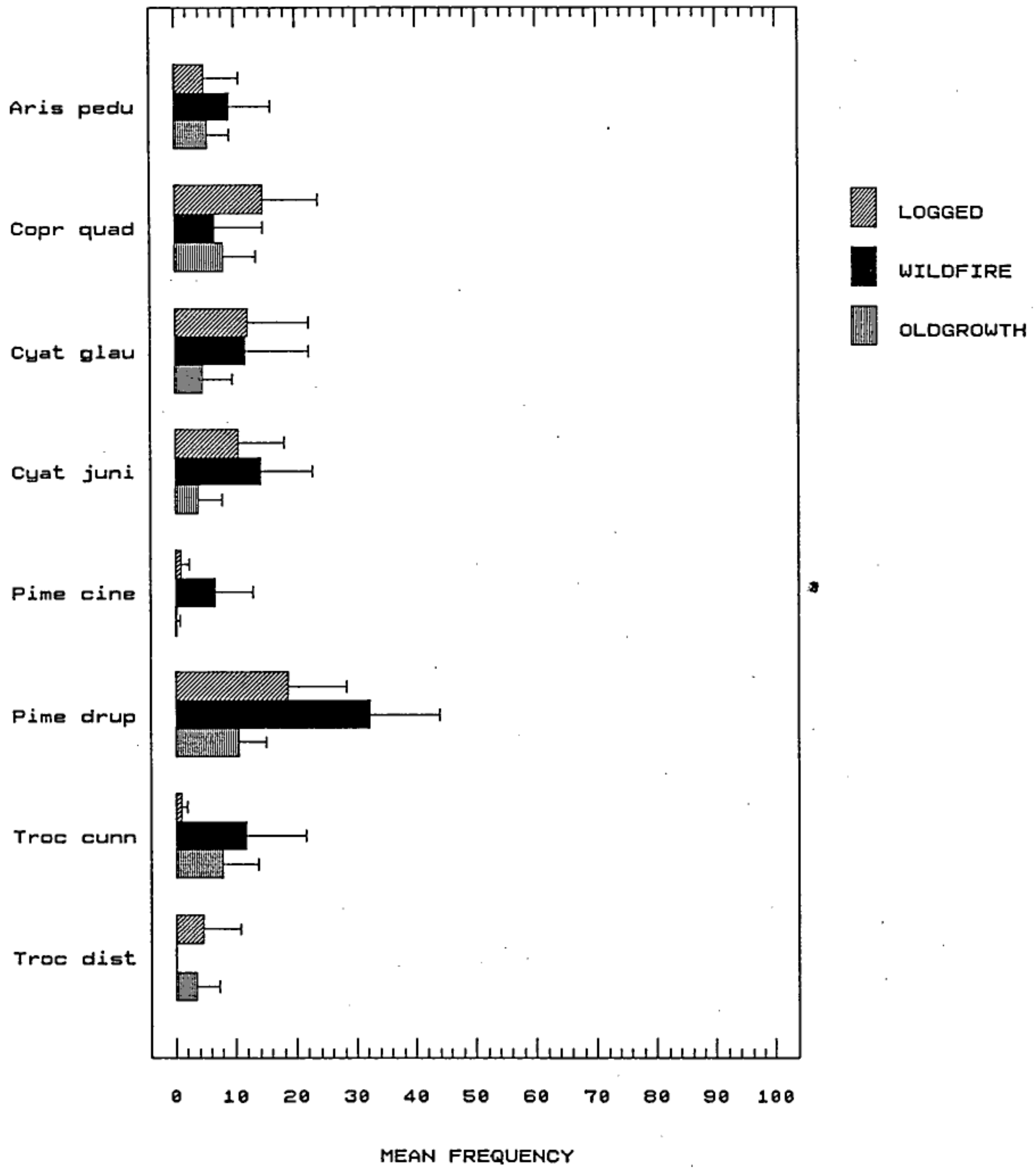


Fig. 3.5(d). Mean frequency of herb and sedge species in silvicultural regeneration (= logged), wildfire regeneration (= wildfire) and oldgrowth mixed forest. (see Appendix 5 for a key to abbreviated species names)

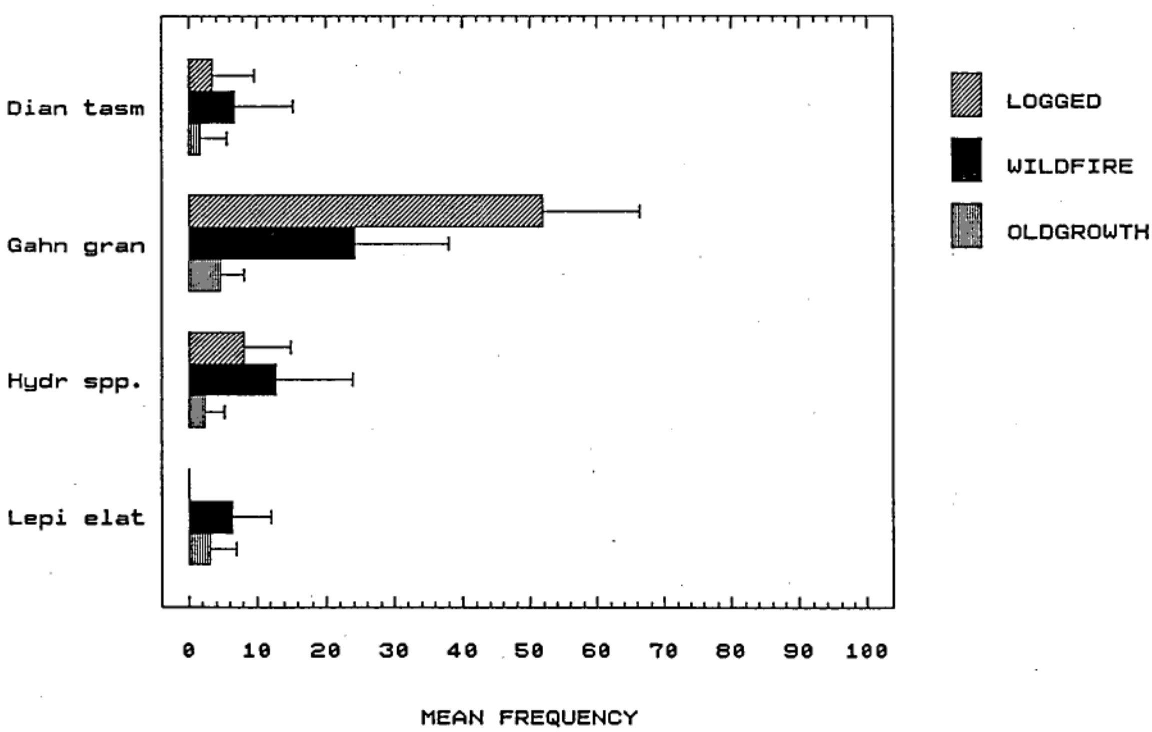


Fig. 3.5(e). Mean frequency of epiphytic species in silvicultural regeneration (= logged), wildfire regeneration (= wildfire) and oldgrowth mixed forest. (see Appendix 5 for a key to abbreviated species names)

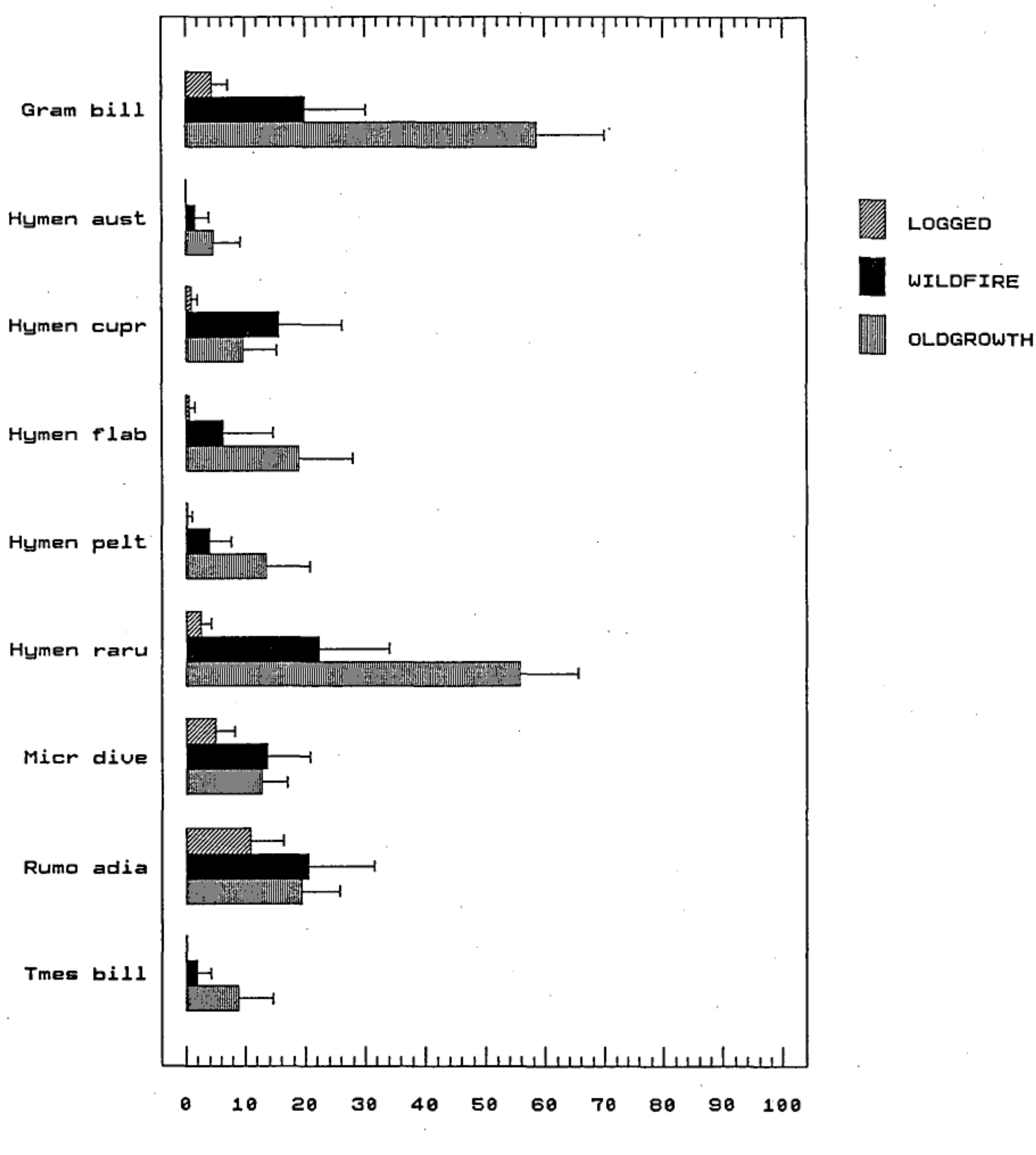


Fig. 3.5(f). Mean frequency of ground fern species in silvicultural regeneration (= logged), wildfire regeneration (= wildfire) and oldgrowth mixed forest. (see Appendix 5 for a key to abbreviated species names)

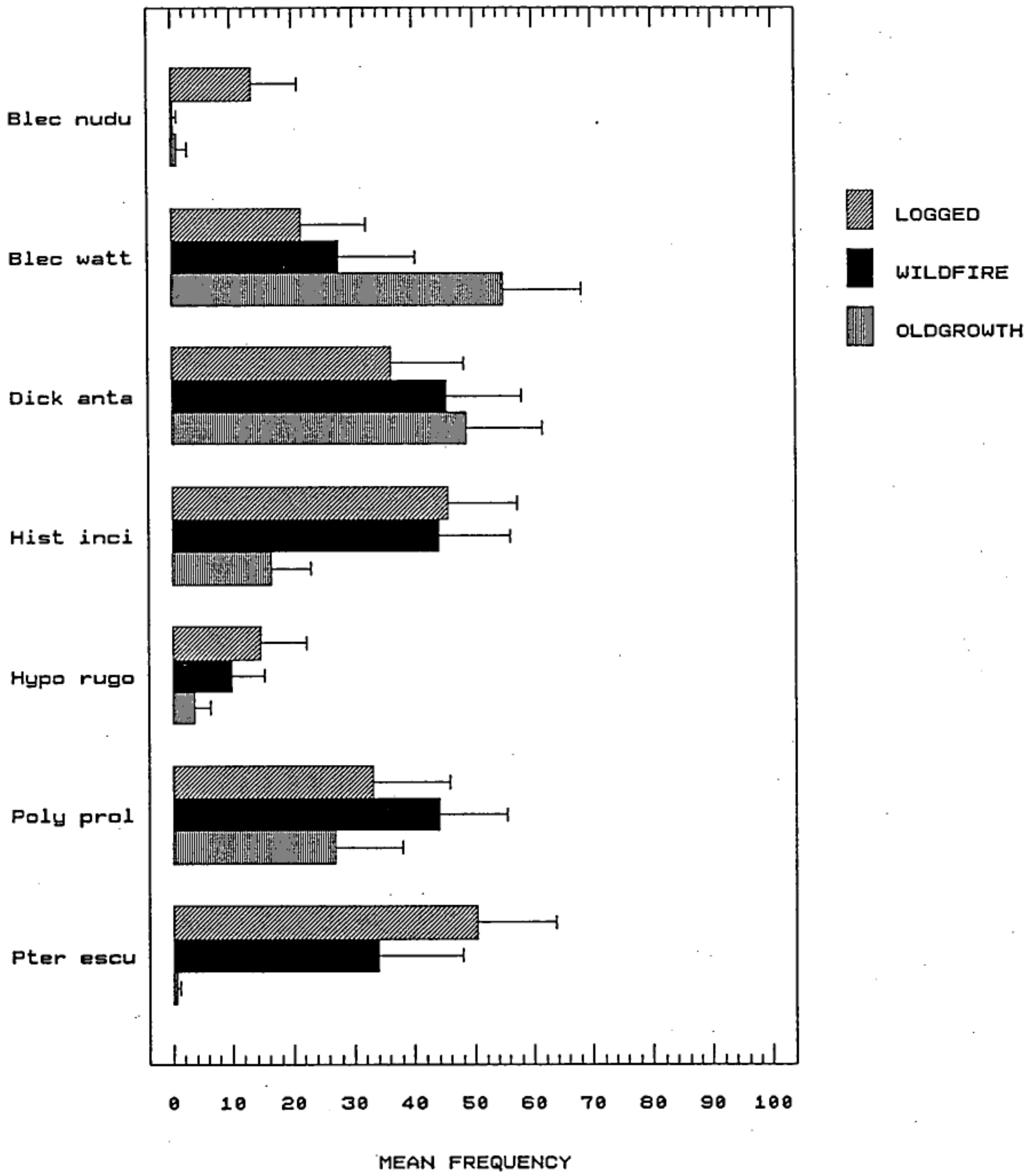
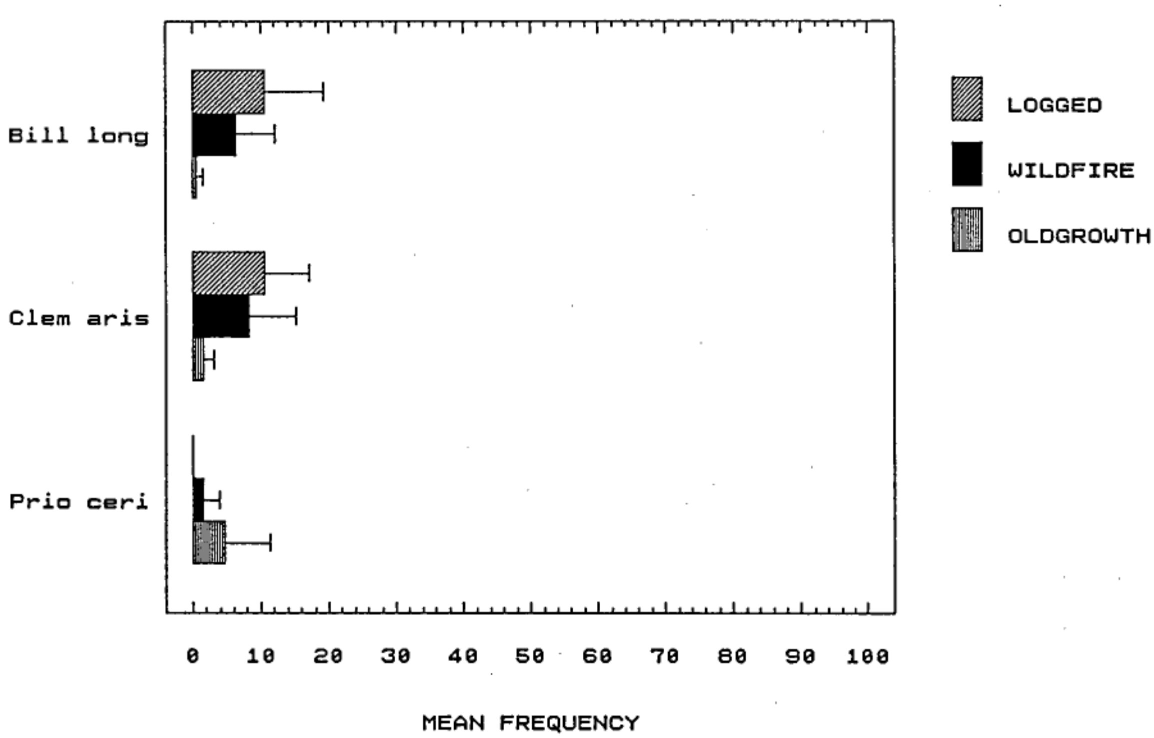


Fig. 3.5(g). Mean frequency of climber species in silvicultural regeneration (= logged), wildfire regeneration (= wildfire) and oldgrowth mixed forest. (see Appendix 5 for a key to abbreviated species names)



Tables 3.9 to 3.15 indicate that 15 rainforest species, four doubtful rainforest species and one non rainforest species had similar frequencies in all site types. Rainforest species in this category appear to be those that: have seed that can be dispersed by vertebrates, e.g. *Phyllocladus aspleniifolius*, *Coprosma quadrifida* and *Cyathodes* spp.; can regenerate vegetatively after fires, e.g. *Cenarrhenes nitida, Anopterus glandulosus;* have a high survival rate after burning, e.g. *Dicksonia antarctica;* or occur at very low frequencies in any of the site types, e.g. *Prionotes cerinthoides.*

Species which are disadvantaged by massive disturbance are those which occur in lower frequencies in one or both of the regenerated site types. Table 3.16 lists species which were most frequent in oldgrowth mixed forest sites and:

A similarly frequent in both silvicultural and wildfire regeneration; or

B significantly more frequent in wildfire regeneration than in silvicultural regeneration.

No common oldgrowth species were more frequent in silvicultural regeneration than wildfire regeneration.

Table 3.16. Species which were most frequent in oldgrowth mixed forest sites and:

A similarly frequent in both silvicultural and wildfire regeneration; or

B more frequent in wildfire regeneration than in silvicultural regeneration.

A	B
Anodopetalum biglandulosum	*Grammitis billardierei*
Atherosperma moschatum	*Hymenophyllum rarum*
Blechnum wattsii	
Eucryphia lucida	
Hymenophyllum australe	
Hymenophyllum flabellatum	
Nothofagus cunninghamii	
Pittosporum bicolor	
Microsorum diversifolium	
Tmesipteris billardierei	

All the species which had significantly lower frequencies in either regeneration type are rainforest species. The ten group A species appear to be affected similarly by silvicultural and natural regeneration processes and include four tree, one tall shrub, one ground fern and four epiphytic fern species. The two group B species are epiphytic ferns and are less frequent after silvicultural treatment than after a wildfire.

Species which are advantaged by massive disturbance are shown in Table 3.17.

Table 3.17. Species which were least frequent in oldgrowth mixed forest sites and:

C similarly frequent in both silvicultural and wildfire regeneration;

D more frequent in wildfire regeneration than in silvicultural regeneration, or

E more frequent in silvicultural regeneration than wildfire regeneration.

C	D	E
Acacia dealbata	*Pimelea cinerea*	*Acacia melanoxylon*
Acacia verticillata	*Pimelea drupacea*	*Acacia riceana*
Billardierei longiflora	*Prostanthera lasianthos*	*Gahnia grandis*
Clematis aristata		*Leptospermum scoparium*
Eucalyptus delegatensis		*Pteridium esculentum*
Eucalyptus obliqua		
Histiopteris incisa		
Hypolepis rugosula		
Monotoca glauca		
Phebalium squameum		
Pomaderris apetala		

The 11 group C species which were present equally after either silvicultural treatment or a wildfire included four rainforest species, three doubtful rainforest species and four non rainforest species. The three group D species that were more frequent after wildfire included two rainforest and one doubtful rainforest species. The five group E species that were more frequent after silvicultural treatment included *Acacia melanoxylon, Gahnia grandis* and *Leptospermum scoparium* which are species often associated with poorly drained sites. It is possible that the habitat of these species would be expanded by the presence of compacted snig tracks in logged areas. Williamson (1990) reported that *Gahnia grandis* is a prominent early colonizer of primary snig tracks in wet forests.

Floristic classification

The TWINSPAN analysis was truncated after three levels to divide the sites into eight clusters. The clusters are listed in Table 3.18. Most of the oldgrowth sites fell into clusters 1 and 2. The first cluster appeared to group sites with a thamnic rainforest understorey while the second cluster contained sites with a callidendrous understorey with some sclerophyllous elements. Clusters 3-8 consisted mainly of regenerated sites. Clusters 3 and 4 contained similar amounts of logged and wildfire sites. Clusters 6 and 8 contained only logged sites while all other sites contained at least two site types. The general failure of the analysis to discriminate between logged and wildfire sites gives some support to the notion that the two sites types are floristically similar, at least when the presence or absence of species are considered.

Table 3. 18. Clusters of sites after TWINSPAN analysis.

cluster	site number	(n) sites	site types	major preferential species
1	4,8,12,23-25,37 39,63,65,69	11	10 oldgrowth 1 logged	*Anodopetalum biglandulosum*
2	2,3,11,16,17,18 21,22,27,33,34 35,38,42-51,57 59,64,68,70-75 78,80,82,83,84 95,96,97,102,103	44	25 oldgrowth 12 wildfire 7 logged	*Acacia melanoxylon, Olearia argophylla, Pomaderris apetala, Pimelea drupacea, Dicksonia antarctica, Polystichum proliferum*
3	5,13,15,19,20,26 36,40,41,54,55 56,58,60,61,62 67	17	9 logged 7 wildfire 1 oldgrowth	*Acacia melanoxylon, Acacia mucronata, Leptospermum scoparium, Melaleuca squarrosa, Blechnum nudum*
4	1,6,7,9,10,14,30 31,32,66,79,93	12	7 wildfire 5 logged	*Acacia dealbata, Atherosperma moschatum, Cyathodes glauca*
5	81,85,92,98,99	5	3 logged 2 oldgrowth	*Atherosperma moschatum, Nothofagus cunninghamii, Dicksonia antarctica, Pimelea drupacea*
6	52,88,91	3	3 logged	*nil*
7	76,86,94,100,101	5	4 wildfire 1 logged	*nil*
8	87,89,90	3	3 logged	*Blechnum nudum*

Ordination

Ordination of the mean species cover abundance data for the 100 sites was carried out in both two and three dimensions. A minimum stress of 0.2439 was achieved in two dimensions and 0.1610 in three dimensions. The stress level is a measure of the goodness-of-fit between the ordination distances and the dissimilarities of the sites. Only the two dimensional solution is considered here, primarily due to the ease of portrayal, and is shown in Fig. 3.6. The distances between the sites are proportional to the dissimilarity in mean species cover abundance between the sites. The 38 oldgrowth sites form a distinct group which can be enclosed within a polygon which excludes the other site types. The logged and wildfire sites are intermingled and cannot be separated easily into two exclusive groups.

A second ordination was carried out on the regenerated sites alone to examine whether there was some separation according to site types in the absence of the oldgrowth sites. The ordination is shown in Fig. 3.7 and indicates no clear separation of site types. This suggests there are not major differences between the vascular plant floristics of silvicultural regeneration and regeneration which results from a wildfire in oldgrowth mixed forest.

The vectors created by the environmental site variables were fitted into the ordinations shown in Figs 3.6 and 3.7. The results are summarised in Table 3.19 which lists the variables, the maximum correlation that could be found between values of the variable and the site scores in each ordination, and the significance level of the correlation. Fig. 3.8 shows the relationship between the environmental site variables which had significant correlations with the ordination scores.

Table 3.19. Results of fitting vectors of maximum correlation for site variables into the ordinations of all 100 sites and the 62 regenerated sites.

Variable	100 sites		62 regenerated sites	
	Maximum correlation	Probability	Maximum correlation	Probability
SITETYPE	0.73	<0.001	0.14	N.S.
ALTITUDE	0.65	<0.001	0.63	<0.001
SLOPE	0.45	<0.001	0.40	<0.001
FERTILITY	0.36	<0.001	0.52	<0.001
ANRAIN	0.33	<0.001	0.28	<0.05
ASPECT	0.17	N.S.	0.15	N.S.
DRYMON	0.07	N.S.	0.17	<0.05
AGE	-	-	0.10	N.S.

The vector analysis of the ordination of the three site types is dominated by the variable SITETYPE which had a greater influence on the floristics of the sites than all other site variables. However, when only the regenerated sites were considered, SITETYPE was only weakly correlated with site floristics and had less influence than any other site variables with the exception of AGE. AGE was not expected to be an influential variable as all the regenerated sites were between 18 and 30 years old. Fig. 3.8 indicates that there is very little relationship between SITETYPE and the environmental site variables. * The vector analyses indicate that the floristic composition of mixed forest is profoundly influenced by either clearfelling and burning or a wildfire but the floristic composition of 20-30-year-old regeneration is more influenced by environmental site variables than by the nature of the disturbance which initiated the regeneration.

* This provides support for the assertion that the environmental variation between the logged, wildfire and old growth sites was similar.

Fig. 3.6. Ordination of 100 sites. (1 = logged, 2 = wildfire, 3 = oldgrowth).

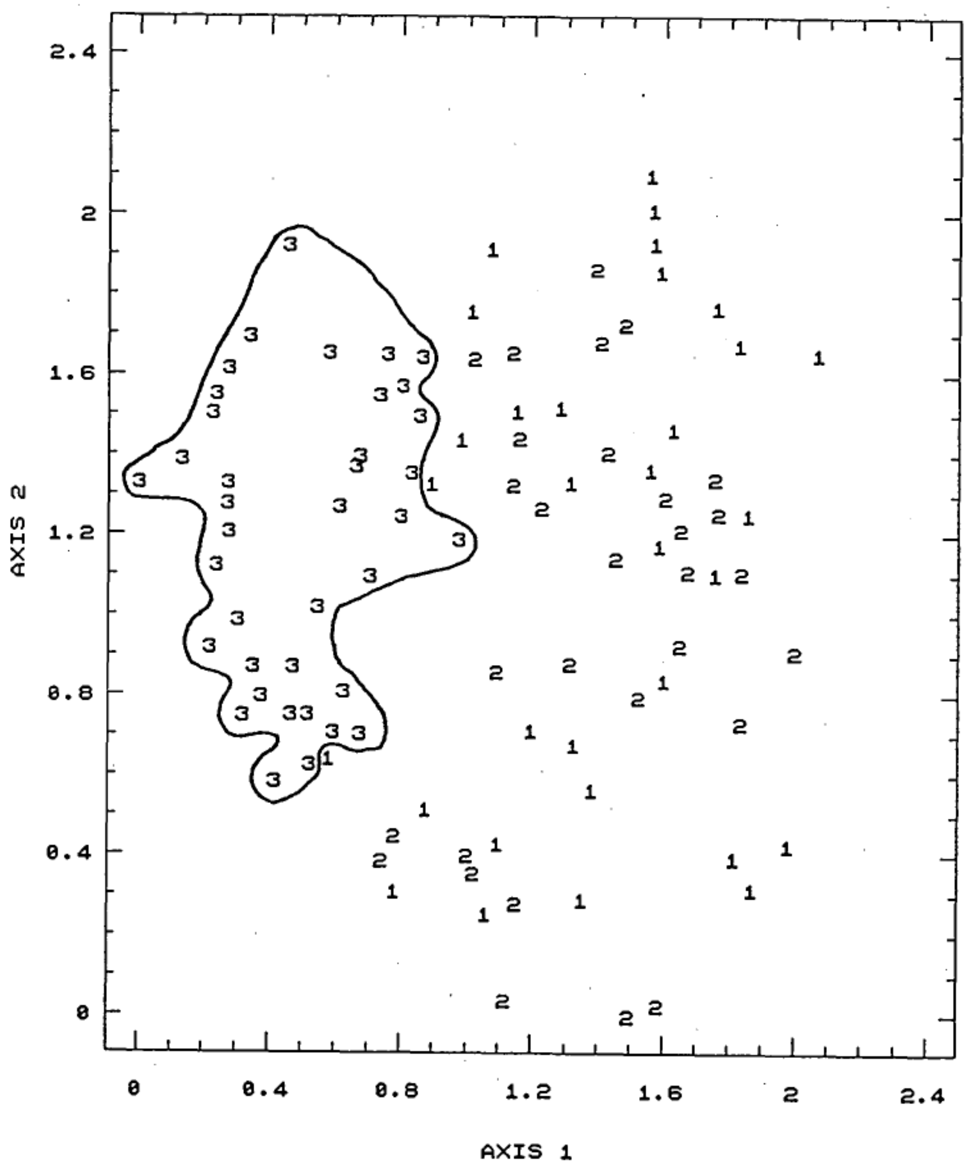


Fig. 3.7. Ordination of 62 regenerated sites. (1 = logged, 2 = wildfire).

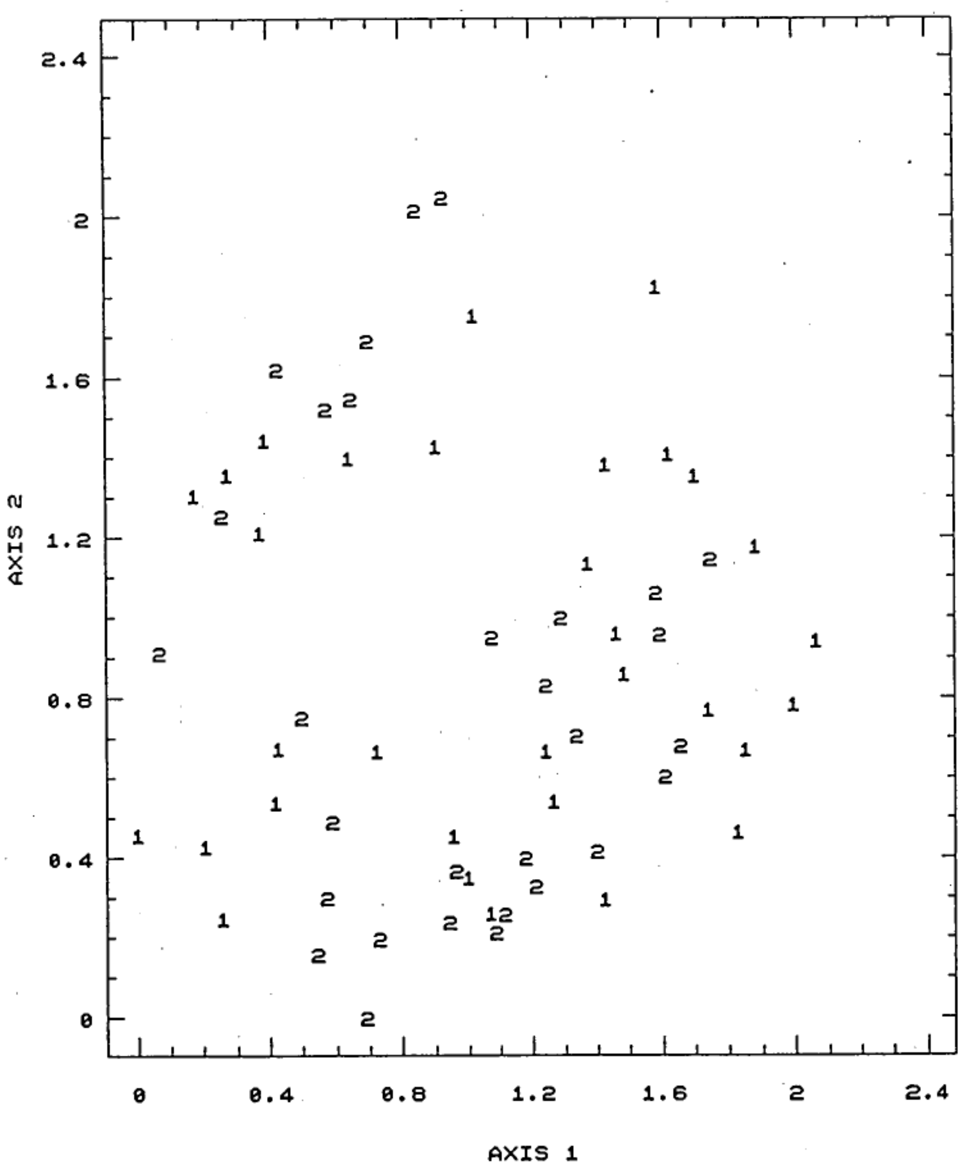
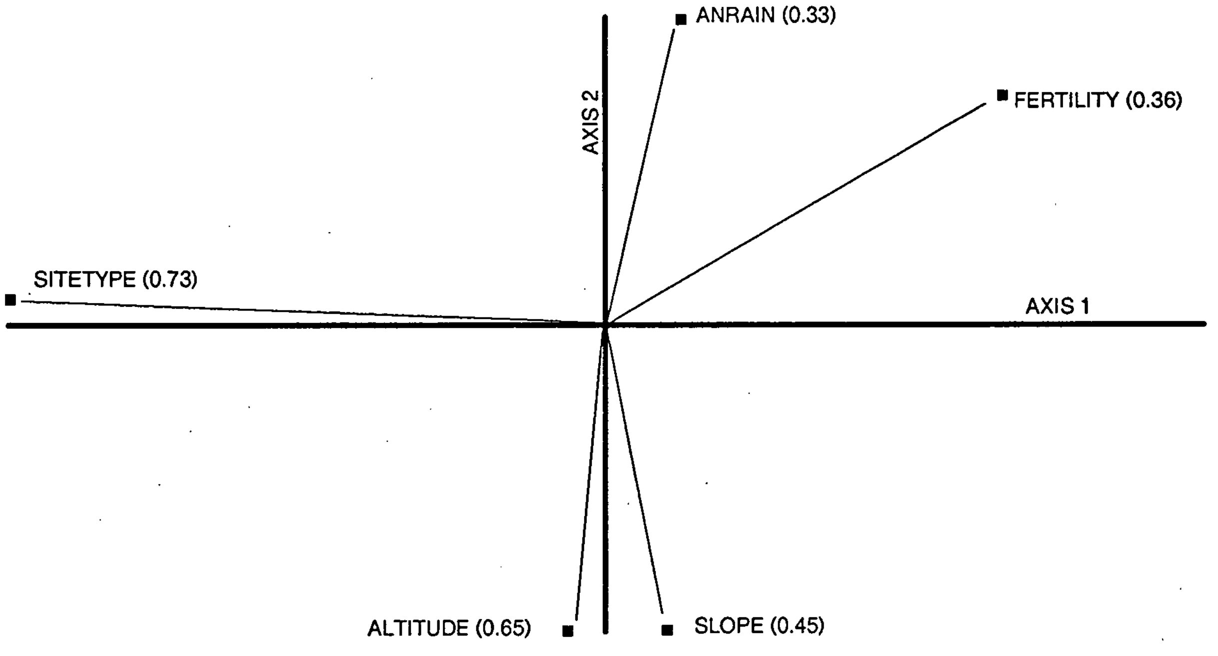


Fig. 3.8. Vectors for environmental site variables with significant correlations with ordination scores for the 100 sites.



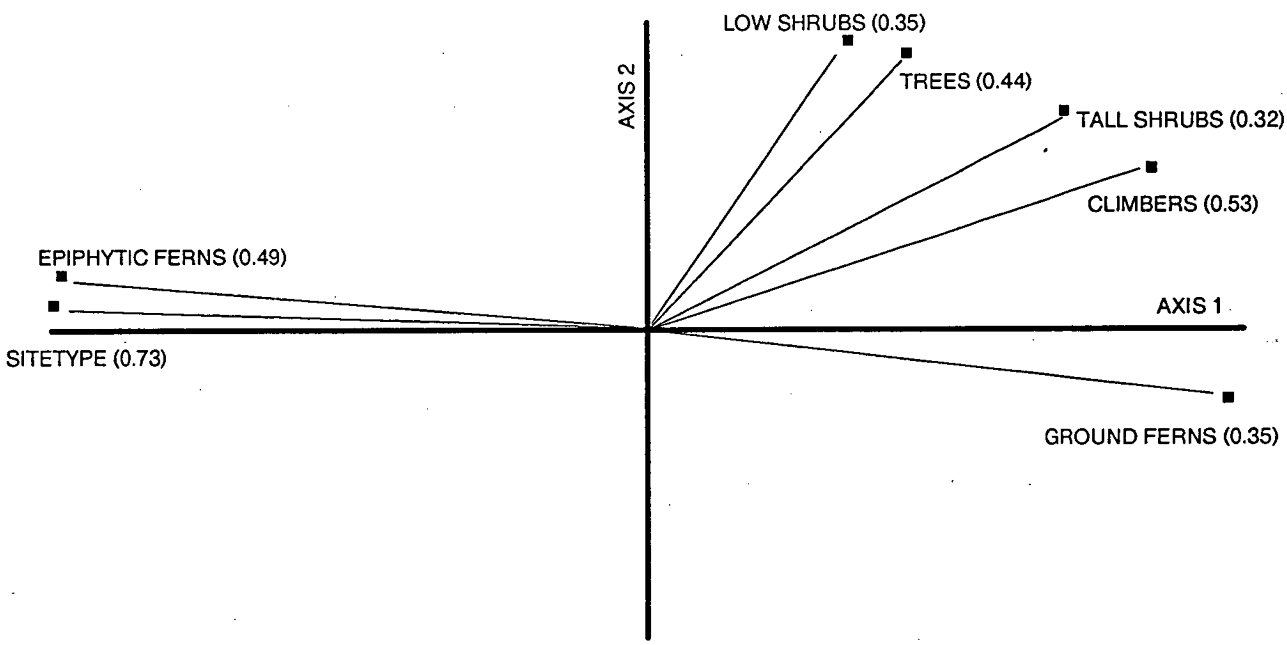
The relationship between the variable SITETYPE and the variables for species richness of tree, tall shrub, low shrub, epiphytic fern, ground fern and climber species was examined by observing their correlations with the ordination scores for the 100 sites. The correlations and their significance are shown in Table 3.20. Fig. 3.9 shows the relationship between the vectors of maximum correlation for the variables.

Table 3.20. Results of fitting vectors of maximum correlation for SITETYPE and species richness variables into the ordinations of all 100 sites.

Variable	Description	100 sites	
		Maximum correlation	Probability
SITETYPE		0.73	<0.001
EPFERN	Epiphytic fern species richness	0.49	<0.001
TREE	Tree species richness	0.45	<0.001
CLIMBER	Climber species richness	0.38	<0.001
FERN	Ground fern species richness	0.36	<0.001
LOWSHRUB	Low shrub species richness	0.35	<0.05
TALLSHRUB	Tall shrub species richness	0.32	<0.001

The species richness variable with the highest correlation with the ordination scores was EPFERN (epiphytic fern species richness). Fig. 3.9 indicates a very strong relationship between site type, epiphytic fern species richness and ground fern species richness. These results complement those derived from the univariate examination of site type and species richness presented in Fig. 3.4. Both analyses indicate that epiphytic fern species richness is least in silvicultural regeneration and highest in oldgrowth forests. The converse is true for ground fern species richness. Fig. 3.9 shows that the species richness for the other life-forms was only weakly associated with site type. The weakest association was with low shrub species richness.

Fig. 3.9. Vectors for species richness variables with significant correlations with ordination scores for the 100 sites.



Discussion

The results of this study indicated that the floristics of oldgrowth mixed forest differ markedly from 20-30-year-old regeneration which resulted from either a wildfire or clearfelling and burning. However, many common species in oldgrowth mixed forest were represented in approximately similar amounts in both regeneration types. Most rainforest species were still present in regenerated forest although they were inconspicuous due to the abundance of sclerophyllous trees and shrubs. This result is consistent with that of Barker (1991) who reported regeneration of most vascular rainforest species within a sclerophyllous scrub that had resulted from a large wildfire in rainforest eight years earlier. Except for epiphytic ferns, species richness was greater for all life-forms in regenerated sites than in oldgrowth forests . Franklin (1992) notes this is a common result in young regeneration, particularly prior to canopy closure, and the diversity is due to a mixture of surviving forest species and weedy pioneer species. The studies of Dickinson and Kirkpatrick (1987) and Tanjung (1992) provide Tasmanian examples of this trend for regenerated wet forests following clearfelling and burning.

There were some significant differences between the floristics of the two regeneration types in the present study. These included a much reduced frequency of the two most common oldgrowth epiphytic fern species in silvicultural regeneration and an increase in frequency of two tree, one tall shrub, one sedge and a fern species which are often associated with disturbed areas. This suggests that the effect of clearfelling and burning a mixed forest differs from the effect of a wildfire. This is not surprising as burning a standing live forest is likely to result in a mosaic of fire intensities which can allow sensitive species such as epiphytic ferns to survive in local refugia. Even a totally fire-killed forest provides much more shade, and many more micro-habitats, than the relatively uniform ash-bed created by high intensity regeneration burns.

Lindenmayer *et al.* (1990) noted differences, particularly in terms of the numbers of hollow-bearing trees, between wildfire and clearfelling on the structure of wet eucalypt forest in Victoria. The Forest and Timber Inquiry (Resource Assessment Commission 1992) considered the degree to which clearfelling and burning was equivalent to natural fire events and noted the following differences: disturbance due to clearfelling operations takes several weeks or months, whereas wildfires can cover the same area in hours; in many forest types trees can resprout after wildfire whereas clearfelling removes most trees; clearfelling results in greater soil disturbance and loss of nutrients than wildfires. Franklin (1992) indicates that most natural disturbances, including wildfire, leave higher 'biological legacies' of surviving trees, dead stags and down logs than most clearfelling and burning operations.

Despite these differences, the results of this study support the hypothesis that an understorey of similar species composition to that of undisturbed mixed forest could eventually develop following a single occurrence of logging, burning and artificial regeneration. Conversely, this study does not support the inference of Taplin *et al.* (1992) that a single occurrence of clearfelling, burning and artificial regeneration of mixed forest will eliminate rainforest understorey species. However, it is probable that successive treatments with short intervening periods would have this effect.

Three of the six common oldgrowth mixed forest species described by Ough and Ross (1992) as being adversely affected by clearfelling, *Microsorum diversifolium, Nothofagus cunninghamii*, and *Pittosporum bicolor* were also affected to a similar degree by a single wildfire. *Grammitis billardierei* was found to be more adversely affected by clearfelling than by a wildfire. The other two species, *Dicksonia antarctica* and *Polystichum proliferum,* occurred in similar frequencies in all three site types in this study.

It is not known whether there will be a recovery of epiphytic ferns as micro-habitats develop in older-aged regrowth up to the planned rotation age of 80 to 100 years, or if there will be sufficient amounts of robust and sexually mature rainforest trees and shrubs for the rainforest element to be maintained in the second rotation. Some 20-30-year-old sites in this study already showed evidence of secondary regeneration of rainforest trees species. *Nothofagus cunninghamii* sprouts from a stump had seeded at site 89 and young seedlings had established. *Atherosperma moschatum* sprouts were flowering and very small seedlings were observed at sites 73 and 74 while *Eucryphia lucida* saplings had sparse flowers at site 55 although no young seedlings were found.

From the information gathered so far, it seems that after a single occurrence of logging, burning and artificial regeneration, the vascular plant floristics are sufficiently similar to the natural regeneration of mixed forest to assume that, in the absence of further logging or fires, the silvicultural regeneration could follow the successional pathways described by Gilbert (1959) and Jackson (1968) and become mixed forest and eventually rainforest. Further work is required to determine whether regrowth mixed forest can be logged at 80 to 100 years and still retain sufficient rainforest elements to return eventually to mixed forest within the life span of the dominant eucalypts.

This study has shown that clearfelling and burning has a significantly greater impact than wildfire on the occurrence of two common epiphytic ferns. It is debatable whether this modification to the natural condition is significant. One possibility is that there will be a subsequent recovery of epiphytic ferns in silviculturally regenerated forests as suitable microsites develop prior to the next disturbance. Even if there is not, there may be suitable habitats available in retained streamside vegetation nearby. Alternatively, the modification could be significant as epiphytic ferns may be representatives of a suite of specialised taxa which includes lichens and bryophytes that may be progressively eliminated from sites after successive treatments. Kantvilas (1990) noted that particular lichen species are the best indicators of forest integrity as they are frequently the first to be eliminated and the last to re-establish in the regenerating forest.

4. A COMPARISON OF STRUCTURE BETWEEN SILVICULTURAL REGENERATION AND WILDFIRE REGENERATION

The results of the previous chapter indicate that the floristics of mixed forest regenerated by clearfelling and burning are, for the most part, similar to mixed forest which has been regenerated by a wildfire. This does not necessarily mean that the structure of the regeneration types is also similar. The structure of each type can be partly described in terms of the growth and density of its dominant trees and tall shrubs. Significant differences in growth and density between the silvicultural and wildfire regeneration would lead to structural differences between the subsequent mature stands. Such differences would indicate that silvicultural treatments have resulted in modified mixed forest regrowth compared to that resulting from the natural regeneration process initiated by wildfire.

Published information on growth and density of young eucalypt stands is very limited although large data sets for major timber species are maintained by the Forestry Commission, Tasmania, in the form of their Continuous Forest Inventory System (Forestry Commission 1985). Ashton (1976) described the diameter and density of young *E. regnans* stands, regenerated by wildfire, in Victoria and reported the relationship between age and diameter as: $\log y = 1.02 \log x$, where y is the eucalypt diameter at breast height (dbh) in centimetres and x is the stand age in years. Diameter increments peaked for stands aged between 40 and 60 years. Density decreased almost exponentially with stand age and ranged from over 200,000 stems in the first year, to about 1000 stems at 26 years (including over 500 suppressed stems) and less than 50 stems by age 220 years. West (1981) used data from repeated measurements of about 105 plots in eucalypt regrowth in Tasmania to develop a model to predict future diameter increment and mortality for 20-80-year-old unthinned stands of *Eucalyptus obliqua, E. regnans* and *E. globulus*. Some information on densities of understorey species in even-aged young eucalypt forest exists (e.g. Cunningham and Cremer 1965) but information on their growth is virtually non-existent despite the fact that six species are commonly used for sawlogs and a further 27 species, including 13 common mixed forest species, can be used for craftwood (Mesibov 1983).

This chapter compares diameter growth of tree and tall shrub species in stands regenerated after clearfelling and burning with diameter growth in unlogged stands regenerated by a wildfire. Similar comparisons are made for height increments of rainforest canopy species. The densities of tree and shrub species are compared between the different types of regenerated forests, and with oldgrowth mixed forest. The largest diameter specimens of common sawlog and craftwood species recorded in oldgrowth mixed forest sites are also reported. There is no attempt to present a mensurational analysis of the growth of tree and shrub species in mixed forest for the purposes of estimating wood volumes. Simple growth and density measures are used to indicate possible structural differences between the two regeneration types and to provide a guide to the sizes of trees which may be available to prospective harvesters of regenerated and oldgrowth mixed forest.

Methods

Mean dominant diameter increments in regenerated stands

The diameters at breast height (1.3 m) of the largest individuals of each tree and tall shrub species were measured with a diameter tape at each 5 m by 5 m plot within a set of ten plots at the 32 logged, 30 wildfire and 38 oldgrowth mixed forest sites described in Chapter 3. Where seedlings had not attained a height of 1.3 m they were recorded as having a notional diameter at breast height of 0.1 cm. The plot data for regenerated sites were averaged to give a mean dominant diameter for each species at each site. The mean diameters were converted to mean annual diameter increments by dividing site means by stand age. A multi-factor analysis of variance procedure was used to test the hypotheses that: (a) the annual mean diameter increments were the same for each species; and (b) that the mean increments were the same for logged sites and wildfire sites. Mean dominant diameter increments were calculated for species which occurred on at least five sites. Sites with zero values for a species were excluded from the calculation.

The mean dominant diameters for individual sites were plotted against stand age to calculate simple linear regression equations for each species of the form $A = xD$ where A = stand age (years) and D = mean dominant diameter at breast height (dbh). The coefficient, x, provides a simple index of the growth of species in regenerated mixed forest stands up to 30 years old. The regression for each species was forced through the origin, i.e. it was assumed that the stand age was zero when the mean dominant diameter was zero. R^2 statistics were calculated for each regression to indicate the amount of variation in age which can be attributed to variation in diameter.

Height increments of canopy species in regenerated stands

The height of the tallest individual of each rainforest canopy species, *Atherosperma moschatum, Eucryphia lucida, Nothofagus cunninghamii* and *Phyllocladus aspleniifolius,* and of *Acacia melanoxylon* was recorded for each 5 m by 5 m plot within sets of 10 plots at the 32 logged and 30 wildfire sites. Saplings less than 2 m tall were measured with a tape. Taller saplings were estimated by eye to the nearest 0.5 m. These methods were impractical for the dominant eucalypts which were usually at least 20 m tall. The height of the tallest eucalypt tree at, or adjacent to, each site was measured using an altimeter where a reasonable line of sight was available. In other cases eucalypt heights were estimated by eye. Heights of other potential canopy species such as *Acacia dealbata* were not recorded. Mean annual dominant height increments were calculated by dividing site means by stand age. A multi-factor analysis of variance procedure was used for the height increment data on rainforest canopy species and *Acacia melanoxylon* to test the hypotheses that: (a) the mean annual dominant height increments were the same for each species; and (b) that the increments were the same for logged sites and wildfire sites. The eucalypt height data were used to provide indicative annual height increments for common eucalypt species in young lowland mixed forest.

Comparison of density of trees and tall shrubs

A stem count was made for all tree and shrub species on each 5 m by 5 m plot within sets of ten plots at the 32 logged, 30 wildfire and 38 oldgrowth mixed forest sites. A mean density was determined for each site and used to calculate the mean, median and maximum density of trees and tall shrubs for each site type. Mean densities were compared between site types using histograms and ANOVAs. Species with mean frequencies of less than 5 percent in all of the site types were excluded from the comparison. The ANOVAs were based on transformed data in order to stabilise the variances between sites. The Log transformation was used such that TDensity = Log(Density+1), where TDensity is the transformed density score. The term (Density+1) was used to prevent undefined values for zero scores. Cochrans C test was used to test for homogeneity of variances for species between site types. Species with heterogeneous variances were excluded from further analysis. A multiple-range test was used for the remaining species to indicate those mean densities significantly different at the 95 percent confidence level.

Sizes of trees and tall shrubs in oldgrowth mixed forest

The data from 380 plots from 38 oldgrowth sites were examined to determine the mean and maximum diameter attained for each commercial sawlog or craftwood species. Species which occurred at less than five oldgrowth sites were excluded from the analysis. The value of the upper quartile was also calculated to give prospective harvesters a guide to the size of timber which may be available. Mean values are not necessarily a useful guide for rainforest species which regenerate in the absence of major disturbance and therefore can occur in a large range of sizes in undisturbed forest.

Results

Diameter increments

Fig. 4.1. shows scatterplots of stand age versus mean dominant diameter for a eucalypt tree species (*Eucalyptus obliqua)*, sclerophyllous shrub species (*Pomaderris apetala)*, rainforest tree species (*Nothofagus cunninghamii*) and a rainforest shrub species (*Tasmannia lanceolata*). The graphs indicate a poor relationship between mean dominant diameter and age. The data are limited to stands aged from 18 to 30 years and there is a marked clumping of data points for 25-year-old stands. However, for the purpose of comparing relative growth, it was assumed that the relationship between diameter and stand age could be described as linear, at least for stands up to 30 years old. Growth is clearly fastest for the eucalypt species and slowest for the rainforest shrub species.

The results of the ANOVA for the mean annual dominant diameter increment data for tree and tall shrub species in regenerated mixed forest are shown in Table 4.1. Factors included in the analysis were species and site type, i.e. logged versus wildfire.

Fig. 4.1. Scatterplots of stand age versus mean dominant diameter for a eucalypt tree species (*Eucalyptus obliqua)*, sclerophyllous shrub species (*Pomaderris apetala)*, rainforest tree species (*Nothofagus cunninghamii*) and a rainforest shrub species (*Tasmannia lanceolata*).

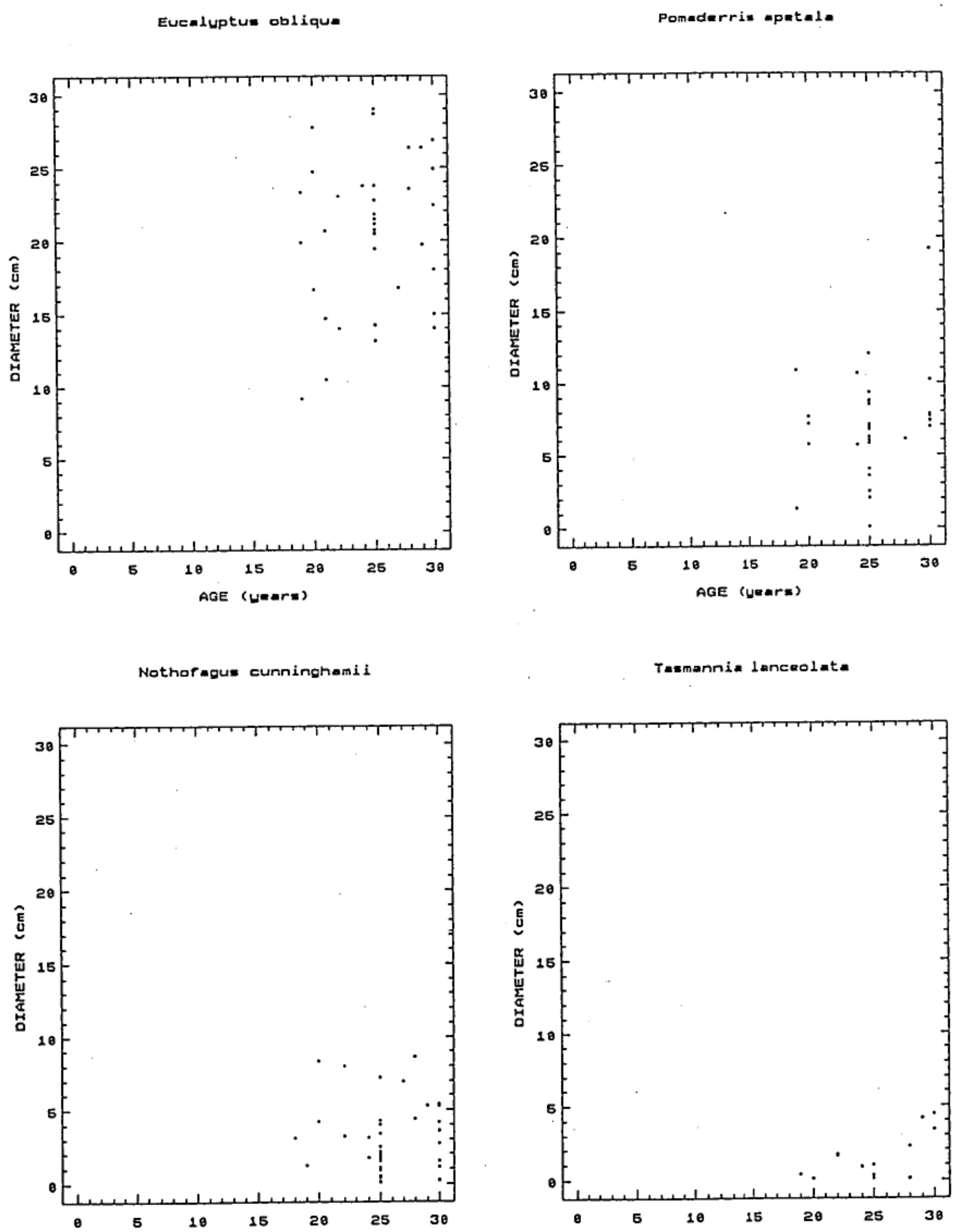


Table 4.1. ANOVA for testing difference in mean annual dominant diameter increments due to species and site type. (d.f. = degrees of freedom)

Source	Sum of Squares	d.f.	Mean square	F-ratio	Sig. level
MAIN EFFECTS					
A: SPECIES	30.330247	20	1.5165123	52.821	0.0000
B: SITE TYPE	0.135508	1	0.1355082	4.720	0.0304
INTERACTIONS					
AB	1.316732	20	0.0658366	2.293	0.0013
RESIDUAL	11.455439	399	0.0287104		
TOTAL (CORRECTED)	43.901096	440			

Table 4.1. shows there is a significant difference ($p<0.001$) between mean dominant increments between species. There is also a significant difference ($P<0.05$) between mean dominant diameter increments between logged sites and wildfire sites. The mean dominant diameter increment was 0.27 cm y^{-1} for logged sites and 0.22 cm y^{-1} for wildfire sites. The interaction between increments for species with site type is also significant ($P<0.05$). Possible reasons for the slightly faster diameter growth of woody species in logged sites may be: greater amounts of fire-killed, but not consumed, debris at wildfire sites which may hamper early growth; and, a deeper ash-bed, hence more rapid growth (Hatch 1960), at logged sites. The difference, although statistically significant, is probably too slight to suggest a structural difference between the two regeneration types.

Table 4.2 lists the mean annual dominant diameter increments for tree and tall shrub species which occurred in at least five sites. Only three of the 21 species have significantly faster growth rates in logged sites than wildfire sites.

Table 4.2. Mean annual dominant diameter increments, and 95 percent confidence intervals, for tree and tall shrub species at logged and wildfire sites. Species shown in bold have means which differ significantly at the 95 percent confidence level between site types.

Species	Logged sites				Wildfire sites			
	n	mean	95% interval		n	mean	95% interval	
			min	max			min	max
Acacia dealbata	**11**	**0.48**	**0.38**	**0.58**	**10**	**0.23**	**0.12**	**0.33**
Acacia melanoxylon	13	0.40	0.31	0.50	11	0.50	0.40	0.60
Anodopetalum biglandulosum	4	0.04	-0.12	0.21	2	0.00	-0.23	0.24
Anopterus glandulosus	3	0.06	-0.13	0.25	5	0.06	-0.09	0.21
Atherosperma moschatum	13	0.13	0.03	0.22	13	0.06	-0.03	0.16
Cenarrhenes nitida	4	0.02	-0.14	0.19	7	0.03	-0.09	0.16
Eucalyptus delegatensis	**6**	**0.83**	**0.69**	**0.96**	**8**	**0.55**	**0.43**	**0.67**
Eucalyptus obliqua	20	0.91	0.84	0.99	21	0.89	0.82	0.96
Eucalyptus regnans	**13**	**0.83**	**0.74**	**0.92**	**10**	**0.47**	**0.36**	**0.57**
Eucryphia lucida	6	0.09	-0.04	0.23	11	0.08	-0.02	0.18
Leptospermum scoparium	8	0.25	0.14	0.37	4	0.22	0.05	0.38
Melaleuca squarrosa	6	0.23	0.10	0.37	2	0.18	-0.06	0.41
Monotoca glauca	19	0.11	0.03	0.19	21	0.09	0.02	0.16
Nothofagus cunninghamii	19	0.18	0.10	0.25	23	0.07	0.00	0.14
Olearia argophylla	10	0.24	0.14	0.35	14	0.27	0.18	0.36
Phebalium squameum	13	0.22	0.13	0.31	17	0.26	0.18	0.34
Phyllocladus aspleniifolius	12	0.05	-0.05	0.14	12	0.04	-0.06	0.13
Pittosporum bicolor	7	0.05	-0.08	0.18	7	0.07	-0.06	0.19
Pomaderris apetala	13	0.28	0.19	0.38	19	0.28	0.20	0.35
Prostanthera lasianthos	1	0.14	-0.19	0.48	5	0.28	0.13	0.42
Tasmannia lanceolata	8	0.05	-0.07	0.17	10	0.04	-0.07	0.14

Table 4.3 shows the mean annual dominant diameter increments for species, irrespective of site type, ranked in increasing order of growth rate. The species are separated into homogeneous groups according to the differences between mean growth rates at the 95 percent confidence level. The table shows that *Eucalyptus obliqua* had the fastest growth followed by *E. delegatensis* and *E. regnans* which had similar growth. The next fastest species were *Acacia melanoxylon* and *A. dealbata*. The remaining species can be separated into two broad groups: a group of sclerophyllous trees and shrubs which includes species of *Pomaderris, Olearia, Phebalium* and *Leptospermum;* and a group of mostly rainforest species including the rainforest canopy species *Atherosperma moschatum, Eucryphia lucida, Nothofagus cunninghamii* and *Phyllocladus aspleniifolius.*

Table 4.3. Separation of species into groups with homogeneous mean annual dominant diameter increments (Mean dbh inc.). Species with an X in the same column form a homogeneous group.

Species	Sample size (n)	Mean dbh inc. cm y^{-1}	Homogeneous Groups									
Anodopetalum biglandulosum	6	0.02	X									
Cenarrhenes nitida	11	0.03	X	X								
Phyllocladus aspleniifolius	24	0.04	X	X								
Tasmannia lanceolata	18	0.04	X	X								
Pittosporum bicolor	14	0.06	X	X	X							
Anopterus glandulosus	8	0.06	X	X	X							
Eucryphia lucida	17	0.09	X	X	X							
Atherosperma moschatum	26	0.10	X	X	X							
Monotoca glauca	40	0.10	X	X	X							
Nothofagus cunninghamii	42	0.12	X	X	X	X						
Melaleuca squarrosa	8	0.21	X		X	X	X					
Prostanthera lasianthos	6	0.21	X	X	X	X	X					
Leptospermum scoparium	12	0.24				X	X					
Phebalium squameum	30	0.24					X	X				
Olearia argophylla	24	0.26					X	X	X			
Pomaderris apetala	32	0.28					X	X	X			
Acacia dealbata	21	0.35					X		X	X		
Acacia melanoxylon	24	0.45								X		
Eucalyptus regnans	23	0.65									X	
Eucalyptus delegatensis	14	0.69									X	
Eucalyptus obliqua	41	0.90										X

Table 4.4 shows the coefficient, significance level, and R^2 statistic for the regression of age on diameter for tree and tall shrub species which occurred in at least five sites. The species are ranked in order of increasing R^2 values. The table shows a significant relationship between age and diameter for all species except *Anodopetalum biglandulosum* which had the slowest diameter growth of all species (see Table 4.3). Rainforest species, in addition to having the slowest growth, show the weakest relationship between age and diameter growth. Presumably this is because they are generally heavily suppressed by larger eucalypts and sclerophyllous shrubs. Growth of rainforest species individuals are likely to be heavily influenced by local stand attributes such as proximity to light gaps. The apparently weak relationship between their age and diameter may also be due to some individuals being considerably younger than the stand age. For example, Table 5.7 in Chapter 5 shows that the age taken for rainforest canopy species to reach a height of 0.1 m in regenerated stands varied from six to ten years. The highest R^2 values were demonstrated by *Eucalyptus, Leptospermum, Phebalium, Melaleuca, Prostanthera* and *Pomaderris* which are all relatively tall sclerophyllous species. Although *Phebalium* and *Pomaderris* have the capacity for subsequent regeneration from soil-stored seed (Cremer and Mount 1965), it is likely that the ages of the dominant individuals of these species, and of *Eucalyptus, Leptospermum, Melaleuca* and *Prostanthera*, are very close to the stand age.

Table 4.4. Coefficient (x), significance level (p-value) and R^2 statistic for the regression of age on diameter for tree and tall shrub species in regenerated forest up to 30 years old.

Species	x	p-value	R^2
Atherosperma moschatum	0.09	<0.01	0.26
Pittosporum bicolor	0.06	<0.05	0.40
Anodopetalum biglandulosum	0.03	N.S.	0.43
Cenarrhenes nitida	0.03	<0.05	0.43
Tasmannia lanceolata	0.05	<0.01	0.47
Phyllocladus aspleniifolius	0.04	<0.001	0.57
Nothofagus cunninghamii	0.11	<0.001	0.60
Eucryphia lucida	0.09	<0.001	0.64
Acacia dealbata	0.36	<0.001	0.65
Olearia argophylla	0.26	<0.001	0.74
Acacia melanoxylon	0.47	<0.001	0.75
Anopterus glandulosus	0.06	<0.01	0.77
Monotoca glauca	0.10	<0.001	0.77
Pomaderris apetala	0.28	<0.001	0.82
Eucalyptus delegatensis	0.67	<0.001	0.83
Prostanthera lasianthos	0.25	<0.01	0.84
Melaleuca squarrosa	0.24	<0.001	0.85
Phebalium squameum	0.25	<0.001	0.85
Eucalyptus regnans	0.69	<0.001	0.86
Leptospermum scoparium	0.24	<0.001	0.89
Eucalyptus obliqua	0.88	<0.001	0.92

Height increments

The results of the ANOVA for the mean annual dominant height increment data for rainforest canopy species and *Acacia melanoxylon* in regenerated mixed forest are shown in Table 4.5. Factors included in the analysis were species and site type.

Table 4.5. ANOVA for testing differences in mean annual dominant height increments for rainforest canopy species and *Acacia melanoxylon* due to species and site type.

Source of variation	Sum of Squares	d.f.	Mean square	F-ratio	Sig. level
MAIN EFFECTS					
A:SPECIES	1.9586390	4	0.4896598	26.067	0.0000
B:SITE TYPE	0.0014365	1	0.0014365	0.076	0.7856
INTERACTIONS					
AB	0.1871673	4	0.0467918	2.491	0.0466
RESIDUAL	2.3105475	123	0.0187849		
TOTAL (CORRECTED)	4.4384368	132			

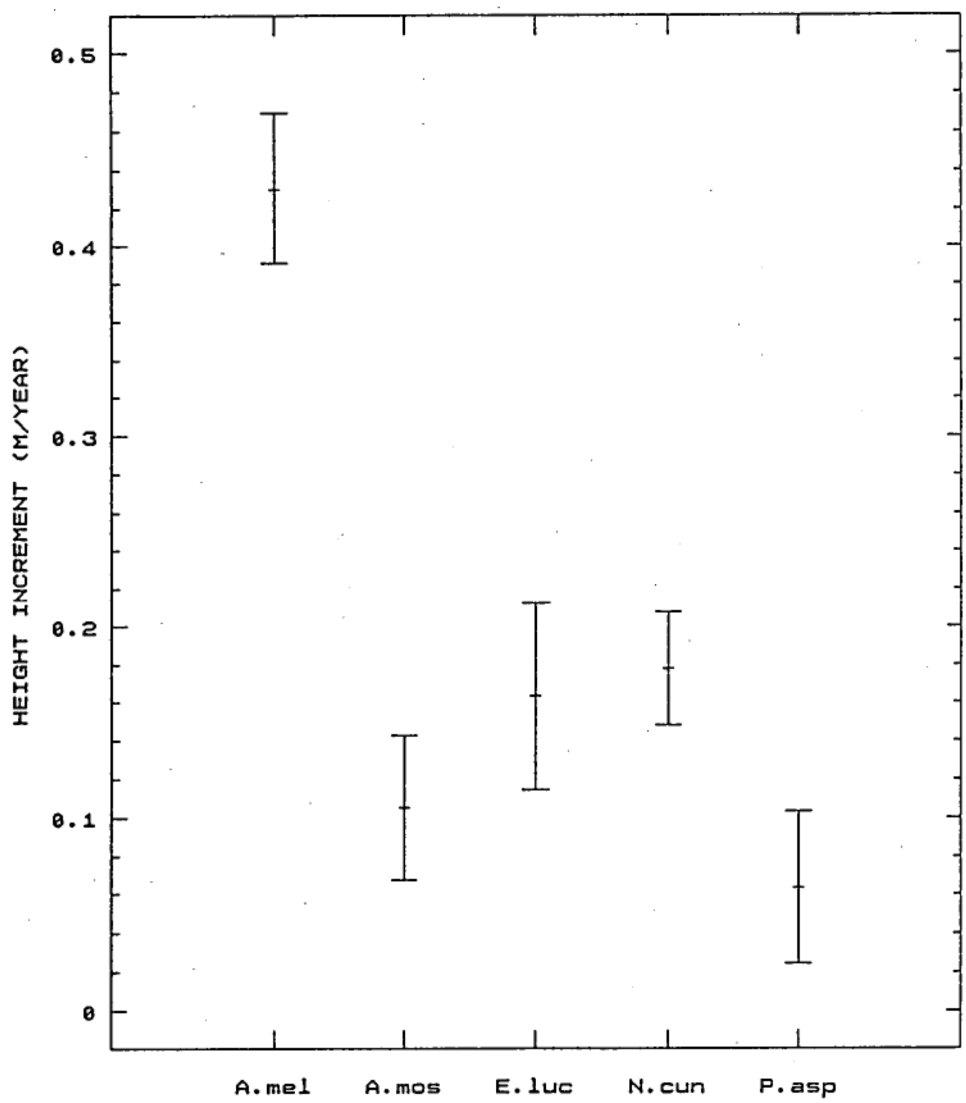
Table 4.5 shows a significant difference ($p<0.001$) between mean annual dominant height increments between species. The differences in means for each site type were not significant and none of the species had significantly different mean height increments between site types.

Table 4.6 shows the mean annual dominant height increments for rainforest canopy species and *Acacia melanoxylon* ranked in increasing order of growth. The species are separated into homogeneous groups according to the differences between mean annual dominant height increments at the 95 percent confidence level. The table shows *Acacia melanoxylon* had much faster height growth than the rainforest canopy species. Of the latter, *Phyllocladus aspleniifolius* had the slowest height growth at only 6 cm y^{-1}. *Nothofagus cunninghamii* had the fastest height growth, at 18 cm y^{-1}, although it was not significantly different from *Eucryphia lucida*. The relationship between the means is shown graphically in Fig. 4.2. The approximate height increments for eucalypt species were 0.8 m y^{-1} for *Eucalyptus delegatensis* and 0.9 m y^{-1} for *E. obliqua* and *E. regnans.* From these data it is predicted that a typical 30-year-old regenerated mixed forest will be dominated by eucalypts about 27 m tall. Underneath, or co-occurring with, a dense layer of sclerophyllous shrubs are *Acacia melanoxylon* up to 13 m tall while further below are *Eucryphia lucida* and *Nothofagus cunninghamii* up to 5 m tall, *Atherosperma moschatum* up to 3 m tall and *Phyllocladus aspleniifolius* up to 2 m tall.

Table 4.6. Separation of species into groups with homogeneous mean annual dominant height increments (Mean ht inc.). Species with an x in the same column form a homogeneous group.

Species	Sample size (n)	Mean ht inc. m y^{-1}	Homogeneous Groups			
Phyllocladus aspleniifolius	24	0.06	X			
Atherosperma moschatum	26	0.11	X	X		
Eucryphia lucida	17	0.16		X	X	
Nothofagus cunninghamii	42	0.18			X	
Acacia melanoxylon	24	0.43				X

Fig. 4.2. Mean annual dominant height increments, and 95 percent confidence intervals, for rainforest canopy species and *Acacia melanoxylon*. (A.mel = *Acacia melanoxylon*, A.mos = *Atherosperma moschatum*, E.luc = *Eucryphia lucida*, N.cun = *Nothofagus cunninghamii* and P.asp = *Phyllocladus aspleniifolius*.)



Comparison of density of trees and tall shrubs

The mean, median and maximum density of trees and tall shrubs in each of the site types is shown in Table 4.7. The species with the highest mean density in oldgrowth sites was *Nothofagus cunninghamii* while *Pomaderris apetala* had the highest mean density in both silvicultural and wildfire regeneration. Mean values were influenced by very high scores at some sites and the median density is a better indicator of the average density for each site type. For example, *Pomaderris apetala* had a mean density of 4400 stems ha^{-1} in wildfire sites and a maximum density of 26,000 stems ha^{-1} but the median density was only 100 stems ha^{-1}.

Tables 4.8 and 4.9 show the results of tests for significant density differences between site types for trees and tall shrubs. Fig. 4.6 shows the transformed mean density, and 95 percent confidence intervals (derived from the within-treatment standard errors), for each species. Species which failed the test for homogeneity of variances are not shown. These were mainly species which occurred only sporadically in some of the site types.

Table 4.7. Mean, median and maximum density (stems ha^{-1}) of trees and tall shrubs in silvicultural regeneration, wildfire regeneration and extant oldgrowth mixed forest.

	Silvicultural regeneration			Wildfire regeneration			Oldgrowth		
	mean	median	max.	mean	median	max.	mean	median	max.
Trees									
Acacia dealbata	95	0	760	57	0	520	3	0	120
Acacia melanoxylon	107	0	760	33	0	320	9	0	120
Acacia mucronata	117	0	2880	57	0	960	0	0	0
Acacia riceana	104	0	2240	0	0	0	0	0	0
Acacia verticillata	59	0	1680	59	0	840	0	0	0
Atherosperma moschatum	157	0	3480	193	0	1280	815	460	3640
Eucalyptus brookeriana	50	0	1400	19	0	560	0	0	0
Eucalyptus delegatensis	159	0	2160	444	0	5480	2	0	80
Eucalyptus nitida	0	0	0	21	0	640	0	0	0
Eucalyptus obliqua	582	360	2560	909	300	3880	26	0	200
Eucalyptus ovata	44	0	1400	0	0	0	0	0	0
Eucalyptus regnans	326	0	2040	764	0	8080	14	0	120
Eucryphia lucida	35	0	520	252	0	2920	276	0	2440
Leptospermum lanigerum	61	0	1080	0	0	0	0	0	0
Leptospermum scoparium	671	0	10600	12	0	200	1	0	40
Melaleuca squarrosa	95	0	2160	79	0	2000	4	0	160
Nothofagus cunninghamii	804	120	8120	983	240	6520	980	520	4040
Phyllocladus aspleniifolius	211	0	2800	468	0	3840	80	0	1200
Tall shrubs									
Anodopetalum biglandulosum	80	0	1880	4	0	80	492	0	3360
Anopterus glandulosus	151	0	2480	107	0	1320	316	0	3960
Banksia marginata	1	0	40	1	0	40	0	0	0
Bedfordia salicina	5	0	160	8	0	240	0	0	0
Cassinia aculeata	41	0	1000	8	0	160	0	0	0
Cenarrhenes nitida	56	56	1680	100	100	1520	42	42	520
Hakea lissosperma	0	0	0	3	0	80	0	0	0
Monotoca glauca	272	40	1600	808	80	10800	18	0	360
Notelaea ligustrina	7	0	200	24	0	400	4	0	120
Olearia argophylla	474	0	8040	423	0	3200	37	0	360
Phebalium squameum	210	0	1520	1421	40	26480	18	0	640
Pittosporum bicolor	61	0	1640	119	0	2120	43	40	320
Pomaderris apetala	3007	0	22480	4437	100	26480	27	0	480
Prostanthera lasianthos	1	0	40	48	0	760	0	0	0
Tasmannia lanceolata	132	0	1760	221	0	3960	53	0	440
Zieria arborescens	120	0	3640	64	0	1080	2	0	40

Table 4.8. Significance tests for differences between transformed mean densities of tree species. (1 = rainforest species, 2 = doubtful rainforest species, 3 = non rainforest species). (Sites with the same letter have similar means at the 95% confidence level, N.S.=non significant.)

Species	Rainforest status	p-value (ANOVA)	Comparison of means by site type logged	wildfire	oldgrowth
Atherosperma moschatum	1	<0.001	A	A	B
Eucalyptus delegatensis	3	<0.05	AB	A	B
Eucalyptus obliqua	3	<0.001	AB	A	B
Eucalyptus regnans	3	N.S.			
Eucryphia lucida	1	<0.01	A	AB	B
Nothofagus cunninghamii	1	<0.001	A	A	B
Phyllocladus aspleniifolius	1	N.S.			

Table 4.9. Significance tests for differences between transformed mean densities of tall shrub species.

Species	Rainforest status	p-value (ANOVA)	Comparison of means (multiple range test)		
Anopterus glandulosus	1	N.S.			
Cenarrhenes nitida	1	N.S.			
Monotoca glauca	1	<0.001	A	A	B
Olearia argophylla	1	N.S.			
Pittosporum bicolor	1	N.S.			
Pomaderris apetala	2	<0.001	A	A	B
Tasmannia lanceolata	1	N.S.			
Zieria arborescens	2	N.S.			

Fig. 4.3(a). Mean transformed density of tree species in silvicultural regeneration (= logged), wildfire regeneration (= wildfire) and oldgrowth mixed forest (= oldgrowth). (See Appendix 5 for a key to abbreviated species names.)

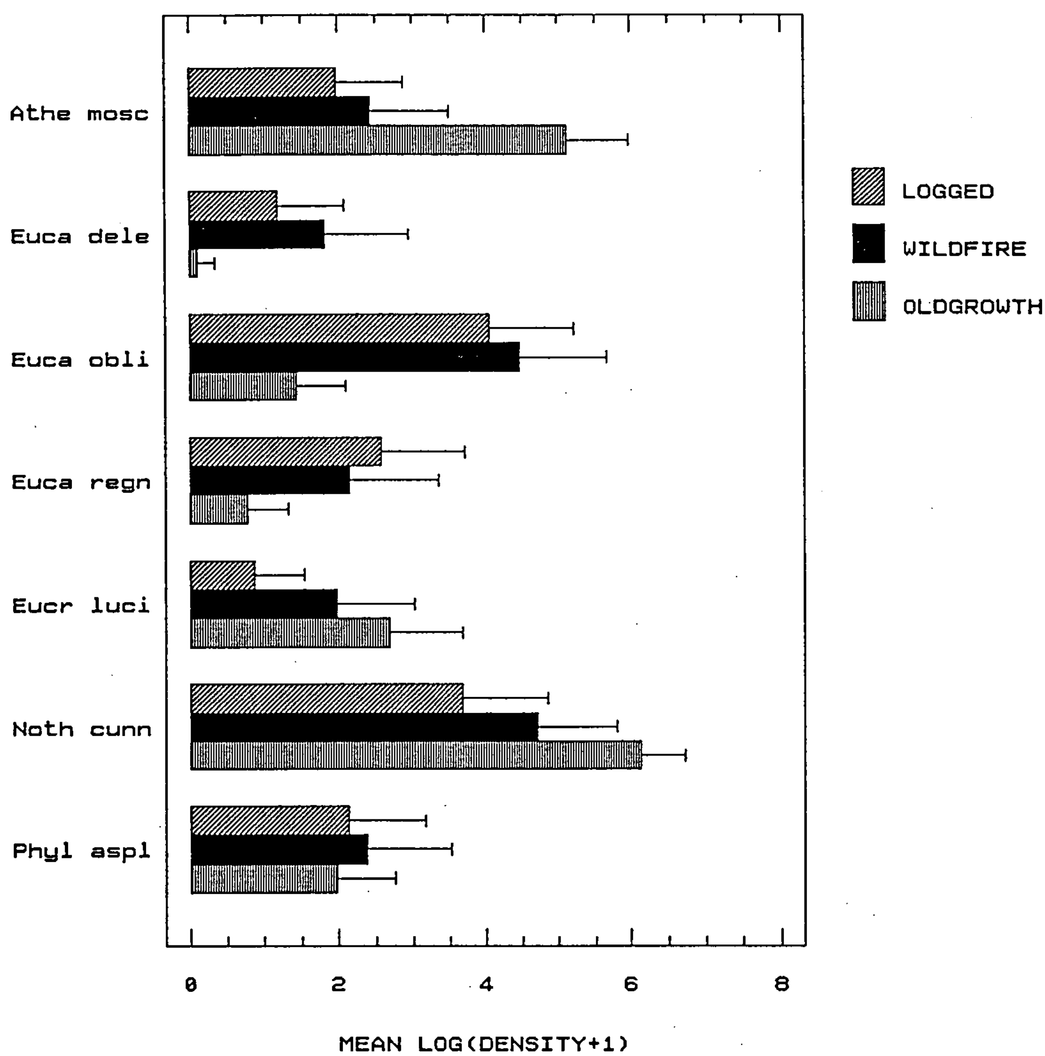
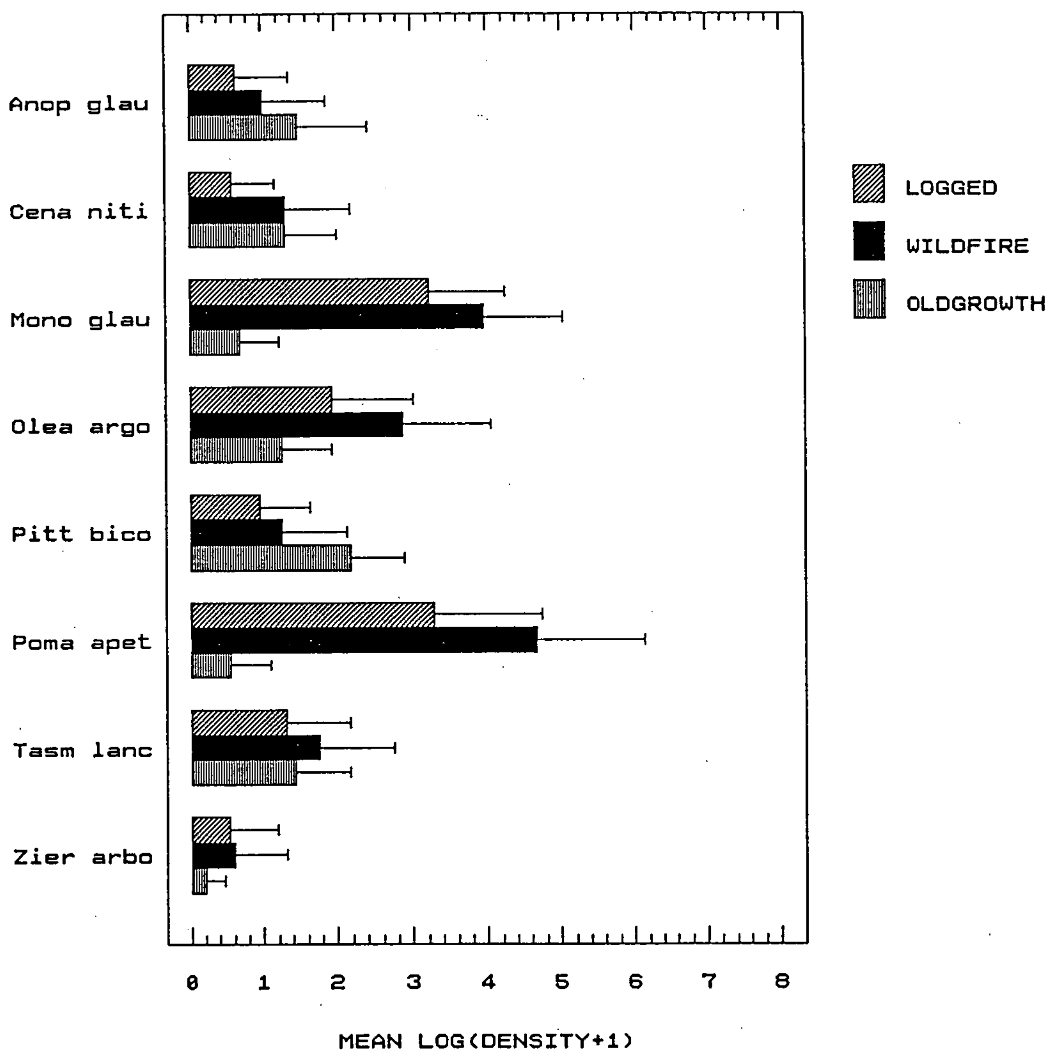


Fig. 4.3(b). Mean transformed density of tall shrub species in silvicultural regeneration (= logged), wildfire regeneration (= wildfire) and extant oldgrowth mixed forest (= oldgrowth). (See Appendix 5 for a key to abbreviated species names.)



The results of the comparison of mean densities for trees and tall shrubs showed a similar trend to the comparison of mean frequencies in Chapter 3. Three of the four rainforest canopy tree species had their lowest density in logged sites but in each case their mean densities were not significantly different from densities in wildfire sites (see Table 4.8). The fourth species, *Phyllocladus aspleniifolius,* had similar densities in all site types. *Eucalyptus delegatensis* and *E. obliqua* both had significantly higher densities in wildfire sites than in oldgrowth sites but there was no significant difference between their density in logged sites and wildfire sites, or between logged sites and oldgrowth sites. There was no significant difference between mean density of *E. regnans* for any of the site types.

Five of the six rainforest shrub species had similar densities in all site types (see Table 4.9). *Monotoca glauca* was least dense in oldgrowth forests but had similar densities in the two regeneration types. Of the sclerophyllous shrubs, *Pomaderris apetala* had much higher densities in regenerated sites than in oldgrowth sites but there was no significant difference between logged and wildfire sites. The density of *Zieria arborescens* was similar for all site types.

Sizes in oldgrowth forest

Table 4.10 lists the mean, maximum and upper quartile diameters recorded for tree and tall shrub species in oldgrowth mixed forest sites. The upper quartile diameters show that several species occur mostly in small size classes. Six species, including *Anopterus glandulosus, Cenarrhenes nitida, Monotoca glauca, Pittosporum bicolor, Phyllocladus aspleniifolius* and *Tasmannia lanceolata*, had upper quartile diameters of less than 10 cm.

Table 4.10. Sample size (n), mean, standard deviation, maximum and upper quartile diameters for tree and tall shrubs in oldgrowth forest.

Life-form	(n)	Mean dbh	Std dev.	Maximum dbh	Upper quartile dbh
		cm		cm	cm
Trees					
Acacia melanoxylon	9	9.3	12.0	37.3	11.7
Atherosperma moschatum	155	15.2	12.7	58.8	22.5
Eucalyptus obliqua	24	179.7	69.5	310.3	208.4
Eucalyptus regnans	12	139.4	80.7	242.5	207.5
Eucryphia lucida	80	20.3	16.8	66.0	33.1
Nothofagus cunninghamii	232	25.3	31.3	194.1	41.5
Phyllocladus aspleniifolius	38	8.1	16.6	67.4	7.2
Tall shrubs					
Anodopetalum biglandulosum	84	10.1	6.6	30.6	12.8
Anopterus glandulosus	48	3.9	3.3	13.9	5.6
Cenarrhenes nitida	21	5.4	5.9	18.0	8.5
Monotoca glauca	10	1.6	2.3	7.0	3.1
Olearia argophylla	21	20.0	13.9	41.8	30.2
Pittosporum bicolor	29	5.6	7.8	37.3	7.9
Pomaderris apetala	12	10.4	5.2	18.0	14.0
Tasmannia lanceolata	29	2.4	3.3	10.9	5.0

Discussion

The results reported here indicate few differences in diameter growth for tree and tall shrub species in logged sites compared with sites regenerated by a wildfire. Small differences were observed for some light-demanding *Eucalyptus* and *Acacia* species which had slightly faster growth on logged sites. These may be attributable to factors such as higher levels of unburnt, or partly burnt debris hampering initial growth on wildfire sites or, alternatively, an enhanced 'ash-bed effect' due to high intensity burning of previously felled vegetation at logged sites. No differences were observed in height growth of rainforest canopy species or *Acacia melanoxylon* between logged and wildfire sites. The absence of growth differences between regenerated site types for most tree and shrub species provides support for the hypothesis that there is little structural difference between the two regeneration types.

Significant growth differences were observed between species. Diameter growth occurred in the following increasing order: rainforest species, sclerophyllous shrub species, *Acacia* species and *Eucalyptus* species. The relationship between age and diameter showed a similar trend. The weak relationship for rainforest species, at least in young mixed forest regeneration, may be due to their occurrence under a dense layer of sclerophyllous trees and shrubs. Variation in growth of rainforest species individuals is probably influenced by local variation in available light.

Read (1985[a]) has noted the potential importance of relative growth rates in determining the canopy composition of rainforest in Tasmania. Seedlings of *Nothofagus cunninghamii* had the highest field growth of the major rainforest canopy species except on poor soils where the growth rates of all canopy species were similar (*ibid*). Hickey (1982) reported field trials of planted fenced seedlings and showed that the species could be ranked in increasing order of growth as: *Phyllocladus aspleniifolius, Atherosperma moschatum, Eucryphia lucida* and *Nothofagus cunninghamii*. The present study indicates that a similar relativity is apparent even for suppressed seedlings in young mixed forest stands. However, growth appears slower in young mixed forest than in similar-aged rainforest. Thus, a mean annual dominant diameter increment for *Nothofagus cunninghamii* of 0.1 cm was recorded in the present study compared to 0.3 cm reported for 20-year-old dense rainforest regrowth by Hickey and Felton (1991). A similar difference exists between reported height increments. Kelly (1989) reported that *Nothofagus* dominants in 9-year-old rainforest regrowth, resulting from logged but unburnt rainforest, grew at 27 to 59 cm y^{-1} compared to an increment of 16 cm y^{-1} reported in the present study.

The reported differences in growth of rainforest species in young mixed forest compared to rainforest regrowth, and to older mixed forest regrowth (see Chapter 6), could be due, at least partly, to the initial presence of a very dense overstorey of sclerophyllous trees and shrubs at the recently regenerated sites. The sclerophyllous species found to grow most densely in the present study was *Pomaderris apetala* which had a mean density of 3000 to 4000 stems ha^{-1} and a maximum density of 26,000 stems ha^{-1} in 20-30-year-old mixed forest stands. Ashton (1976) found that the closely related species *Pomaderris aspera* not only regenerated prolifically after fire but could also regenerate following the degeneration of *Helichrysum* and *Cassinia* understoreys. It attained most of its potential height after 30 years, and density after 40 years, following a wildfire. The species is believed to have a longevity of about 100 years although its numbers decline after about 40 years due to agents such as snow damage, and beetle and fungal attack (*ibid*). It is possible that this dense sclerophyllous shrub layer depresses the growth of rainforest species and inhibits their flowering and subsequent regeneration at the site. It would be interesting to see if an increased growth rate, flowering and subsequent regeneration of rainforest species corresponds to a decline in *Pomaderris.*

The density of *Pomaderris apetala* was similar in both regeneration types in the present study. Most rainforest shrub species had similar densities in all three site types but most of the rainforest tree species had lower densities in regenerated sites although there were no significant differences between logged and wildfire sites. *Monotoca glauca* was an interesting exception in that it occurred in higher densities in regenerated sites although it is considered a rainforest species by Jarman *et al.* (1991).

The results of the growth and density comparisons undertaken here indicate there is little, if any, difference between the stand dynamics of silvicultural and wildfire regeneration of mixed forest. However, Lindenmayer *et al.* (1990) have demonstrated that there are more hollow-bearing trees, which are remnants of the previous stand, in wildfire regeneration of wet eucalypt forest than in silvicultural regeneration. It was certainly apparent in the present study that there were far more dead standing eucalypts in 20-30-year-old wildfire regeneration than in silvicultural regeneration. This structural difference between the two regeneration types probably has greater implications for faunal habitat, particularly for arboreal mammals, than on the habitat of vascular plants.

5. MECHANISMS FOR RAINFOREST SPECIES REGENERATION FOLLOWING WILDFIRE OR CLEARFELLING AND BURNING

Massive disturbance of lowland mixed forest occurs mainly as wildfire or as clearfelling, regeneration burning and artificial application of *Eucalyptus* seed. These disturbances are more likely to adversely affect rainforest species than sclerophyllous species which are generally better adapted to survive, or regenerate after, fire (Cunningham and Cremer 1965). Gilbert (1959) reported that woody rainforest species can regenerate immediately after a wildfire in undisturbed mixed forest and are not reliant on subsequent invasion from outside seed sources. Cremer and Mount (1965) reported a similar result for silviculturally regenerated areas although they observed that *Atherosperma* did not regenerate initially because it appeared to need shade to re-establish.

Harper (1977) summarises the forest regeneration strategies after disasters as:

- regeneration from locally produced seed (which usually requires that the disaster occurs soon after an act of seed dispersal, or that species have persistent seeds that are stored in the soil).
- regeneration from seed dispersed from elsewhere.
- regeneration from dormant buds which produce basal shoots.
- regeneration from a reserve of established seedlings.

The last strategy is effective for some disasters such as windstorms or diseases of mature individuals but is obviously ineffective after fire.

The species which will be least able to recolonize disturbed mixed forest sites in the medium term are those which either do not regenerate from soil-stored or canopy-stored seed, cannot be transported over long distances by wind or vertebrates, rarely (or never) regenerate from sprouts, or require particular micro-sites for recolonisation which are rarely found in young regrowth forests.

Hickey *et al.* (1982) indicate the availability of locally produced seed for three of the four rainforest canopy species which occur in mixed forest. Peak seedfall of *Atherosperma moschatum*, *Eucryphia lucida* and *Nothofagus cunninghamii* occurs about February at low elevations. This roughly corresponds with the time at which most wildfires occur and is about one month earlier than when most regeneration burns are carried out. Heavy seedfalls of *N. cunninghamii* occur every second or third year with very little seed produced in intervening years. Seedfall of *A. moschatum* and *E. lucida* is consistent each year. *Phyllocladus aspleniifolius* is reported to have mast seeding years (Read 1989) with seedfall occurring about February (Barker 1992).

There are no published studies on soil seed banks of oldgrowth mixed forest in Tasmania although Howard (1974) and Neyland (1991) carried out studies of the seed banks under rainforest by monitoring germination from soil samples in a glasshouse. Howard (1974) found that viable seeds of woody species were poorly represented under an old rainforest canopy although grasses, rushes, sedges and herbs not present in the vegetation were abundant. Neyland (1991) also reported a very small correspondence between the extant vegetation at the sites studied and the seeds present in the soil. Only four out of the 26 species that geminated from soil samples were rainforest species with the remainder being invader species such as thistles, grasses, fireweeds and introduced weeds (*ibid*).

Dispersal mechanisms of seed of some Tasmanian rainforest tree species have been reported by Hickey *et al.* (1982) and Read and Hill (1983). French (1990, 1991) and French and O'Dowd (1992) have studied the potential for vertebrate dispersal of common species in wet sclerophyll forests in south eastern Australia.

Regeneration of rainforest species from sprouts following fire has been reported by Hill and Read (1984) and Barker (1991). Barker lists three rainforest trees and nine rainforest shrubs which can regenerate from sprouts.

This chapter describes studies to determine: the amount and composition of regeneration which resulted from soil samples taken from a subset of sites described in Chapter 3; the frequency of seedling regeneration and sprout regeneration of rainforest canopy species following massive disturbance; the re-establishment time for rainforest canopy species in regenerated mixed forest; and trends in logging coupe sizes to estimate the time required for some common rainforest species to recolonise coupes from surrounding forest. The conclusion at the end of the chapter includes a summary of the reported regeneration mechanisms for common vascular species in Tasmanian lowland mixed forests.

Methods

Seed bank study

A seed bank study was undertaken to determine the role of buried seeds in the recovery of mixed forest after massive disturbance, by observing which species are able to regenerate from soil-stored seed. Germination was monitored over two spring-autumn periods. This period was chosen because the studies of Howard (1974) and Neyland (1991) had been of short duration, i.e. one year and four months respectively.

Soil samples were taken in 1991 from nine of the 38 oldgrowth mixed forest sites described in Chapter 3. Sampling was carried out in late winter-early spring so that most short-lived seed from the previous seedfall would have either germinated, decayed, or been destroyed. The nine sites occurred on seven rock types and are listed in Table 5.1. Five samples were taken from each site by using a sharp spade to extract a 'tile' of soil, approximately 30 cm square by 5 cm deep, from within each of plots 1 to 5 (see Fig. 3.2). The litter layer was scraped from the surface of each tile prior to extraction. The tiles were subsequently broken apart by hand and the soil was sieved through a 6.5 mm mesh to remove roots and other large material. The samples from each site were bulked and thoroughly mixed in a portable concrete mixer. A layer of soil about 2 cm thick was then placed on a 2 cm bed of vermiculite in two 34 by 28 cm germination trays for each site. The volume of soil in the trays was approximately 3.8×10^{-3} m^3 which represents 7.6 percent of the soil contained in a 1 m square and to depth of 5 cm.

The trays were stored in a glasshouse, kept moist with automatic mist sprays and monitored at least monthly for seven months until the end of March 1992. A count of germinants of woody species was made each month. Some germinants subsequently died but were included in a cumulative count for each tray. The presence of non-woody species was noted but not quantified. Some weed species, including *Clematis vitalba*, *Epilobium* spp. and *Solanum* spp., which were common in the vicinity of the glasshouse, occurred in the trays as well as in other soil containers. The germinants were harvested at the end of March 1992 and monthly monitoring did not recommence until September 1992 when subsequent spring germination had begun. The last monitoring was in March 1993.

Table 5.1. Oldgrowth mixed forest sites sampled for ground-stored seed.

Site	PI type	Rock type	Community[1]	Sample date
43	E2dM-	Cambrian greywacke	OB1001	19/8/91
46	E2cM-	Cambrian greywacke	OB101	20/8/91
47	E2dM+	Cambrian greywacke	OB1001	19/8/91
49	E2cTM	Pre-Cambrian sediments (unmetamorphosed)	OB1001	19/8/91
63	E1dSM2T(Ddf)	Triassic non-marine sediments	OB1100	16/7/91
64	E1cM3T	Jurassic dolerite	REG110	26/7/91
65	E1dTM3	Parmeener super group	OBL1100	31/7/91
69	E2cM+	Pre-Cambrian dolomite	NIT1	20/8/91
83	E1bM-	Parmeener super group	REG110	25/9/91

[1] according to the classification of Kirkpatrick *et al.* (1988)

Comparison of seedling/sprout ratios for rainforest canopy species in silvicultural versus wildfire regeneration.

The regeneration mode for the rainforest canopy species *Atherosperma moschatum, Eucryphia lucida, Nothofagus cunninghamii* and *Phyllocladus aspleniifolius* was recorded in each 5 m by 5 m plot for the 62 regenerated sites described in Chapter 3. The recording method was to class the rainforest stems in each plot as being either: 0 = seedling origin; 1 = both seedling and sprout origin; and, 2 = sprout origin. Classes 1 and 2 were combined for analysis. A Chi-squared statistic was calculated to indicate whether the proportion of seedling regeneration for each species was the same for logged plots and plots burnt by a wildfire.

Re-establishment time for rainforest canopy species in regenerated mixed forest.

An examination of stem ages of rainforest canopy species in mixed forest was carried out to determine their re-establishment times and differences, if any, between times for particular species. Stems of the largest individual of *Atherosperma moschatum, Eucryphia lucida, Nothofagus cunninghamii* and *Phyllocladus aspleniifolius* were cut from a subset of the logged and wildfire sites described in Chapter 3.* Stems were not cut in permanent plots for obvious reasons. Stems were cut at stump heights of 0 to 0.2 m. Stem sections were removed and subsequently sanded and their annual rings counted under a magnifying glass or dissecting microscope. Some small individuals were

* It was assumed that the largest stems would normally be the oldest individuals on the plot.

uprooted so that their rings could be accurately counted at a 'stump height' of 0 m. Transverse sections of nine of the smallest seedlings were mounted onto microscope slides. The sections were prepared by boiling 1 cm stem sections for at least one hour to soften the wood prior to cutting thin sections with a razor blade. The origin of regeneration, i.e. seedling or sprout was recorded. It was assumed that growth ring formation was annual. This has been demonstrated for *Nothofagus gunnii* and *Phyllocladus aspleniifolius* in Tasmania (Ogden 1978) and for *Nothofagus solandri* and *Phyllocladus trichomanoides* in New Zealand (Norton *et al.* 1987). A variable called LAG was defined which equals the stand age - the ring count for a stem at a site. Stand age was deemed to be the year of the regeneration burn for logged sites, and the year of the wildfire for wildfire sites. LAG gives an approximate time for re-establishment of rainforest tree species on a site.

Coupe size study

Forestry Commission records were examined to list the size of logging coupes in Geeveston and Smithton Districts for the period from 1960 to 1993. Coupes which were regenerated by wildfire, planting or by retained seed trees were deleted from the analysis as they are departures from normal practice in these districts. The median coupe size was calculated for each district and decade.* The median sizes were compared, using a Kruskal-Wallis ANOVA procedure based on ranks, to indicate if there were spatial and/or temporal trends in the size of logging coupes.

* Median size was used, rather than mean size, as the data indicated that there were a few very large coupes in each district which would result in the mean size being a poor estimator of the average value.

Results and discussion

Seed bank study

Germination results from the two trays for each site were combined and are reported in Appendix 6. The frequency of all standing woody species recorded at each site is also shown to compare the composition of the seed bank with the extant vegetation. Table 5.2 is a summary of woody species which germinated in the seed trays. The minimum period for germinants to appear is shown as well as the mean and maximum density of viable seeds for each species.* Mean viable seed density refers to the average number of germinants recorded from soil samples from the nine sites. Maximum viable seed density was recorded from the site which showed the greatest amount of germination for a particular species. **

* The table also indicates those species that have persistent seeds (defined by Grime (1979) as seed which remains viable in the seed bank for more than one year).

** The seedling viability estimates are derived from glasshouse germination environments and therefore species with special germination requirements, such as temperature extremes, may not be detected.

Table 5.2. Germination of woody species from soil-stored seed.

Woody species	Minimum period for germination (months)	Mean viable seed density (m^{-2})	Maximum viable seed density (m^{-2})	Persistent seeds
Acacia dealbata	1	11.7	105	?
Acacia melanoxylon	2	17.5	140	yes
Acacia spp.	3	4.4	13	yes
Cassinia aculeata	4	8.8	79	no
Cyathodes juniperina	14	42.4	315	yes
Eucryphia lucida	1	4.4	40	no
Gaultheria hispida	17	2.9	26	yes
Monotoca glauca	14	16.1	118	yes
Phebalium squameum	14	11.7	65	yes
Phyllocladus aspleniifolius	4	10.2	52	yes
Pomaderris apetala	4	42.4	381	yes
Sprengelia incarnata	18	1.5	13	yes
Trochocarpa gunnii	17	20.5	158	yes

Table 5.3 summarises the seed bank data for the nine sites and shows the number of woody species which germinated from each sampled site (woody species richness), the Jaccard similarity coefficient for the similarity between woody species in the seed bank and in the standing vegetation, the viable seed density in a 1 by 1 by 0.05 m sample of soil and the total species richness (excluding glasshouse contaminants). The Jaccard coefficient (J) was calculated as follows:

$J = n_{++}/(n_{++}+n_{+-}+n_{-+})$ where n++, n+- and n-+ are calculated from the two-way table below

		Site seed bank: present	Site seed bank: absent
Standing vegetation	present	n_{++}	n_{+-}
	absent	n_{-+}	n_{--}

Table 5.3. Woody species richness, similarity with standing vegetation, density of viable seed and total species richness.

Site	Woody species			Total species
	Richness	Similarity (Jaccard)	Viable seed density	Richness[1]
			(m^{-2})	
43	3	0.13	590	6
46	1	0.10	50	4
47	1	0.00	10	4
49	3	0.14	140	8
63	3	0.11	240	9
64	3	0.00	200	10
65	4	0.17	80	9
69	5	0.27	410	7
83	2	0.17	40	3
mean	2.8	0.12	195.5	6.7

[1] excludes glasshouse contaminants (*Acer* sp., *Clematis vitalba*, *Epilobium* sp. and *Solanum* sp.)

Table 5.~~3~~2 indicates 13 woody species, which represent 11 genera, that are capable of regenerating from soil-stored seed. The inclusion of *Eucryphia lucida* maybe unwarranted as it germinated in low numbers and Hickey *et al.* (1982) found that *Eucryphia lucida* seed stored at ambient temperatures was unlikely to germinate after two years. Grime (1979) draws a distinction between transient and persistent seed banks and defines the former as one in which none of the seed remains in a viable condition for more than one year. If *Eucryphia lucida* is excluded, five of the persistent soil-stored species are rainforest species (*sensu* Jarman *et al.* 1991). They are *Cyathodes juniperina, Gaultheria hispida, Monotoca glauca, Phyllocladus aspleniifolius* and *Trochocarpa gunnii*. Only one of these species, *Gaultheria hispida*, was reported by Howard (1974) or Neyland (1991) as germinating from soil samples, although Read (1989) notes that *Phyllocladus aspleniifolius* appears to be soil-stored for at least 18 months. The epacridaceous species, *Cyathodes juniperina, Monotoca glauca* and *Trochocarpa gunnii*, all took more than a year to germinate, which may explain their absence from the shorter studies of Howard (1974) and Neyland (1991). However, Howard did report germination of three additional woody rainforest species, *Coprosma quadrifida, Pittosporum bicolor* and *Tasmannnia lanceolata,* which were not recorded from soil

samples in the present study. Howard also reported a small amount of germination of *Nothofagus cunninghamii* and attributed this to a few surviving seeds from the summer seedfall a few months previously.

Table 5.3 confirms the poor correspondence between species presence in the standing vegetation and the seed bank. The mean similarity coefficient was only 0.12. In a literature review of seed bank studies of temperate deciduous forests, Pickett and McDonnell (1989) cite numerous studies which conclude that standing vegetation and seed banks differ markedly. They reported that only in repeatedly disturbed arable fields does seed bank composition sometimes coincide with standing vegetation. Table 5.4 shows a comparison of soil seed density and species richness for the present study with a number of seed bank studies in old temperate forests reported in Pickett and McDonnell (1989). The table indicates a similar species richness and seed density to two of the three other forest types studied.

Table 5.4. Comparison of seed density and species richness in seed banks of old temperate forests. (Data from Pickett and McDonnell 1989, * this study.)

Forest type	Viable seed density (woody species) (m^{-2})	Total species richness
110-year-old beech-yellow birch-sugar maple	91	8
>100-year-old Allegheny hardwoods	99	6
c. 100-year-old ash with sycamore	758	15
*oldgrowth mixed forest	196	7

Sprout ratio study

Results of the comparison of the proportion of seedling regeneration in logged plots with that of wildfire plots are shown in Table 5.5.

Table 5.5. Frequency of plots with either seedling-only or sprout regeneration. Note that plots with sprouts may also have seedling regeneration present. (*A. mos = Atherosperma moschatum, E. luc = Eucryphia lucida, N. cun = Nothofagus cunninghamii* and *P. asp = Phyllocladus aspleniifolius)*

Plots with:	Logged sites				Wildfire sites			
	A. mos	*E. luc*	*N. cun*	*P. asp*	*A. mos*	*E. luc*	*N. cun*	*P. asp*
seedling-only	18	11	57	53	38	29	86	62
sprouts (±seedlings)	10	2	23	0	12	6	29	0
proportion (seedling-only)	0.64	0.85	0.72	1.00	0.76	0.83	0.75	1.00

The table indicates that all *Phyllocladus aspleniifolius* regeneration occurred as seedlings. The proportion of seedling-only regeneration was very similar for the other species for both site types. This was confirmed by non significant p-values of 0.27, 0.88 and 0.62 for *Atherosperma moschatum, Eucryphia lucida* and *Nothofagus cunninghamii* respectively for the hypothesis that the proportions were the same across site types. Differences might have been expected as regeneration burns are likely to be more intense than most wildfires and the mechanical disturbance due to logging could damage rootstocks and hence reduce regeneration by sprouting. Apparently these factors had little influence on sprouting ability of the species. The data were combined across site types to test the hypothesis that the proportion of seedling-only regeneration was the same for the three species which sprouted (see Table 5.6).

Table 5.6. Frequency of plots with either seedling-only or sprout (± seedling) regeneration.

Plots with:	Sites		
	A. mos	*E. luc*	*N. cun*
seedling-only	56	40	143
sprouts (±seedlings)	22	8	52
proportion (seedling-only)	0.72	0.83	0.73

The p-value for the hypothesis was 0.3 and indicates that sprouting frequency after massive disturbance is similar for the three species. This does not imply that the sprouting frequency in undisturbed stands is similar. In fact, several authors (e.g. Gilbert 1959, Read and Hill 1985) have noted that *Atherosperma moschatum* frequently regenerates by sprouting whereas *Eucryphia lucida* and *Nothofagus cunninghamii* regenerate mainly as seedlings.

Re-establishment time for rainforest canopy species.

Ring counts from 49 samples are shown in Appendix 7. The age of the sampled stands ranged from 21 to 30 years and the age of the stems ranged from 6 to 30 years. Table 5.7 shows the mean diameter, age, and LAG (the difference between stand age and stem age) according to species, origin (seed or sprout) and stump height. No attempt was made to distinguish between stems from logged sites and wildfire sites.

Table 5.7 indicates that seedlings with higher stump heights had larger diameters and lower LAG intervals. This is an artefact of the field sampling which required larger stems to be cut *in situ* whereas smaller, younger stems were able to be uprooted and sampled *ex-situ* or, at least, cut much closer to ground level. Fig. 5.1 shows the mean LAG interval versus stump height for *Nothofagus cunninghamii* seedlings. The differences are not significant at the 95 percent confidence level.

Table 5.7. Species, number of samples (n), origin, stump height, mean diameter of stump, mean age and mean LAG (stand age-stem age).

Species	n	Origin	Stump ht	Mean diam.	Mean age	Mean LAG
			m	cm	years	years
Atherosperma moschatum	4	seed	0.0	0.5	8.3	17.5
	5	seed	0.1	2.7	16.8	9.8
	2	sprout	0.1	7.0	25.0	0.0
	1	sprout	0.2	11.2	27.0	1.0
	1	sprout	0.3	13.9	23.0	2.0
Eucryphia lucida	1	seed	0.0	1.0	30.0	0.0
	4	seed	0.1	5.0	20.3	6.0
	1	sprout	0.1	7.5	22.0	3.0
Nothofagus cunninghamii	3	seed	0.0	0.9	16.3	10.3
	7	seed	0.1	4.0	18.9	7.7
	4	seed	0.2	10.3	22.8	5.5
	2	sprout	0.1	3.6	24.0	1.0
	3	sprout	0.2	12.8	25.0	2.0
Phyllocladus aspleniifolius	4	seed	0.0	1.2	15.5	11.5
	6	seed	0.1	4.3	19.5	5.8

Fig. 5.2 shows the mean LAG interval for seedlings with a stump height of 0.1 m. The interval ranged from 5.8 years for *Phyllocladus aspleniifolius* to 9.8 years for *Atherosperma moschatum* although there was no significant differences in LAG interval between the species. As *Phyllocladus aspleniifolius* is the only one of the species with soil-stored seed it appears that seedlings of *Atherosperma moschatum, Eucryphia lucida* and *Nothofagus cunninghamii* have arisen from seed deposited a few years after the wildfire or regeneration burn. Alternative explanations could be that growth rates are very slow or that heavy browsing of very small seedlings has delayed the time between seedling germination and height growth to 0.1 m. These species, especially *Atherosperma moschatum,* are susceptible to vertebrate browsing (Hickey 1982). However, browsing is not the sole explanation as the LAG interval for seedlings with a stump height of zero exceeds 10 years for *Atherosperma moschatum, Nothofagus cunninghamii* and *Phyllocladus aspleniifolius* (see Table 5.7).

Fig. 5.3 compares the mean LAG time, regardless of species, for seedlings and sprouts with a stump height of 0.1 m. There is a significant difference between the two with seedlings taking an average of 6.4 years to reach a height of 0.1 m and sprouts requiring only one year to reach the same height.

The results confirm Gilbert's (1959) observation that rainforest species can regenerate, at least from sprouts, soon after wildfire. The seedlings in the present study took a few years to establish and, with the likely exception of *Phyllocladus aspleniifolius,* may have resulted from fresh seed deposited from sources either within the disturbed area or adjacent to it. There were no significant differences between the re-establishment time of the four rainforest canopy species although the youngest seedlings were *Atherosperma moschatum* which lends some support to Cremer and Mount's (1965) observation of delayed re-establishment of *Atherosperma* following clearfelling and burning. Maybe *Atherosperma,* which has very light wind-dispersed seed, is more able than the other rainforest canopy species to occupy subsequent regeneration niches in the developing forest. Read (1985[b]) demonstrated that *Atherosperma* is more tolerant of low light levels than other rainforest canopy species. This may allow it to occupy microsites where it is too dark for other species to survive.

Fig. 5.1. Mean LAG interval and 95 percent confidence limits for *Nothofagus cunninghamii* seedlings at three stump heights.

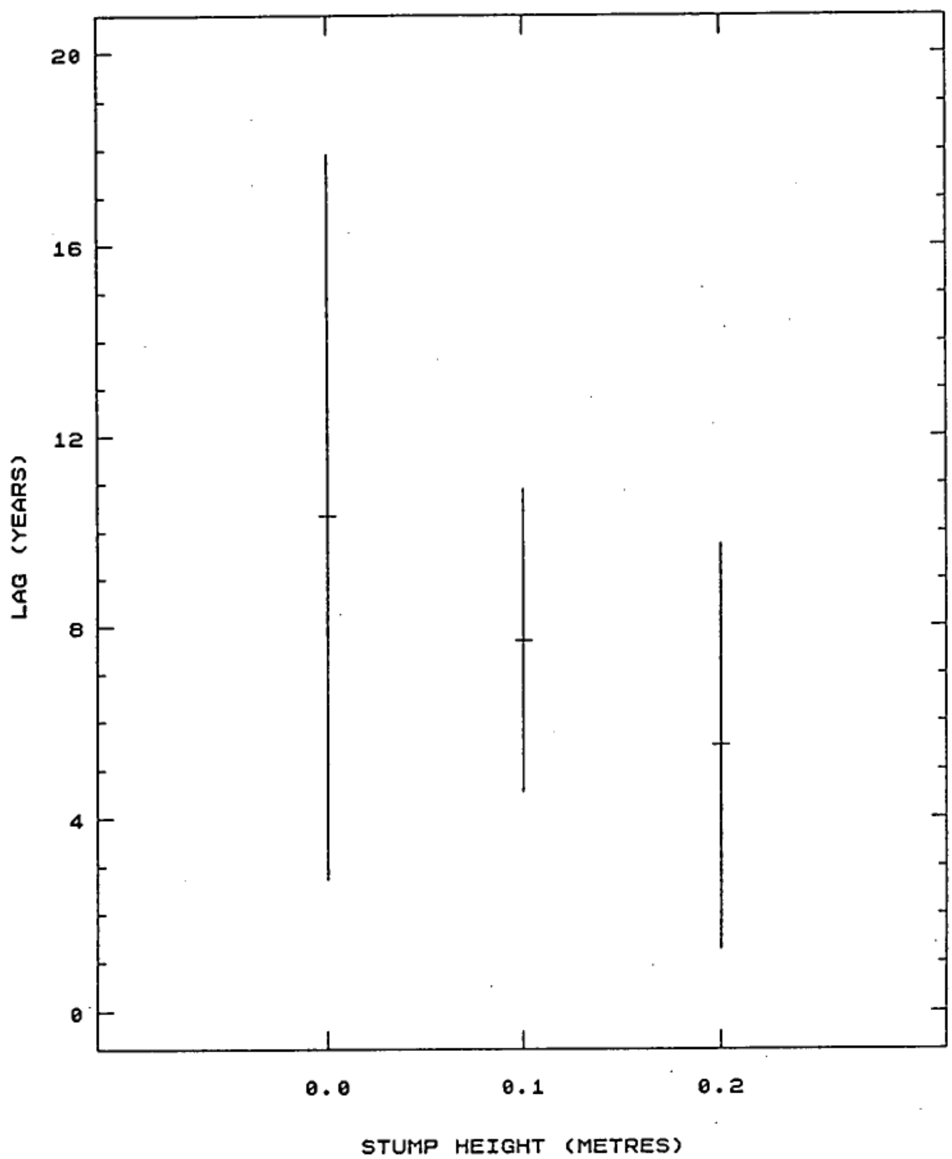


Fig. 5.2. Mean LAG interval and 95 percent confidence limits for *Phyllocladus aspleniifolius* (P.ASP), *Eucryphia lucida* (E.LUC), *Nothofagus cunninghamii* (N.CUN) and *Atherosperma moschatum* (A.MOS) seedlings at a stump height of 0.1 m.

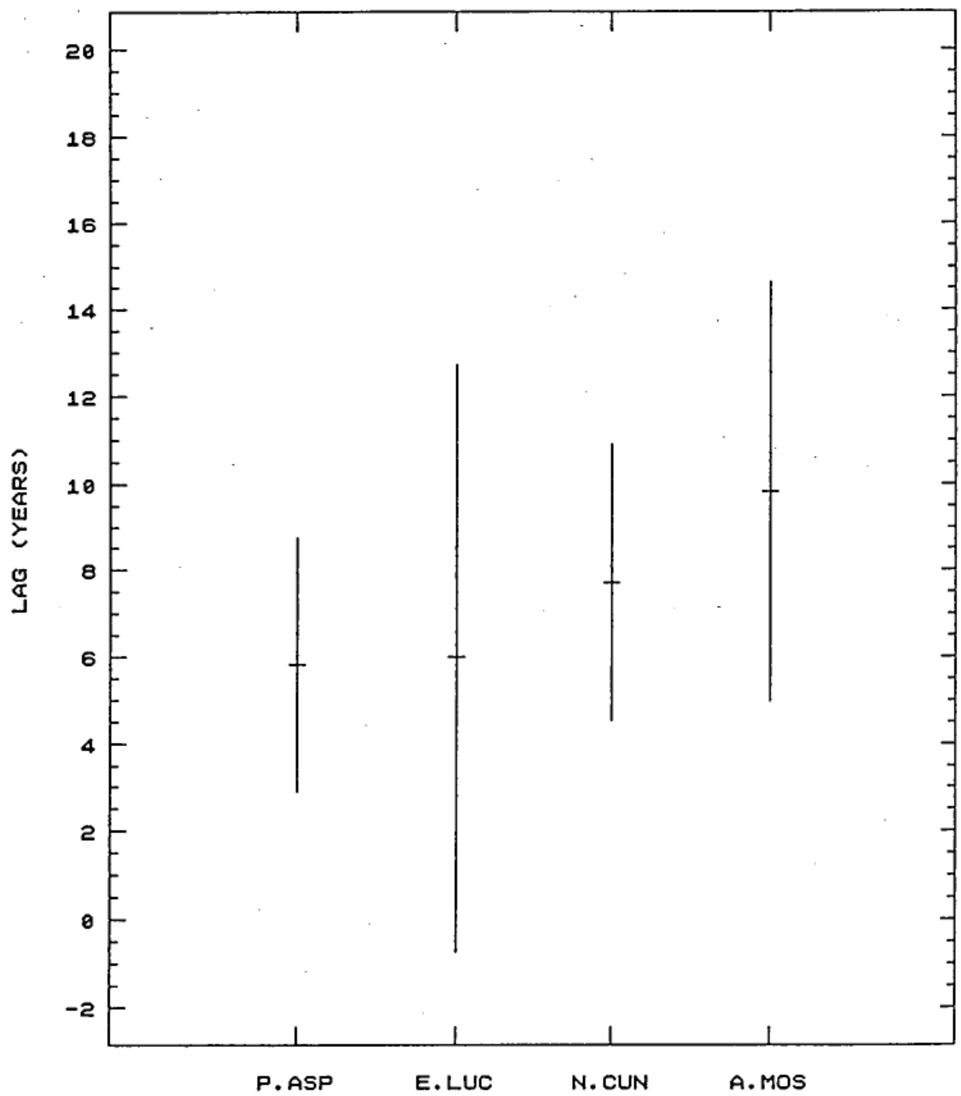
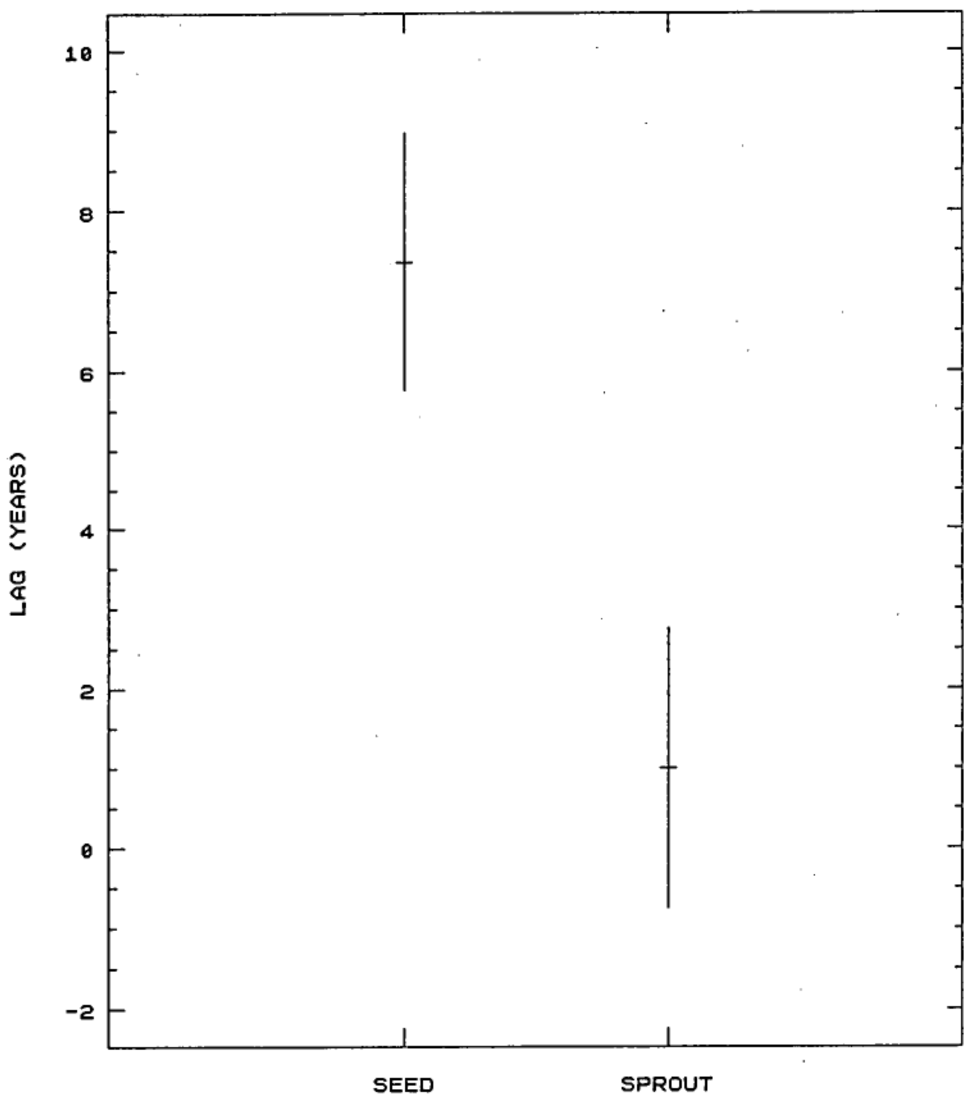


Fig. 5.3. Mean LAG interval and 95 percent confidence limits for seedlings and sprouts, of rainforest canopy species, at a stump height of 0.1 m.

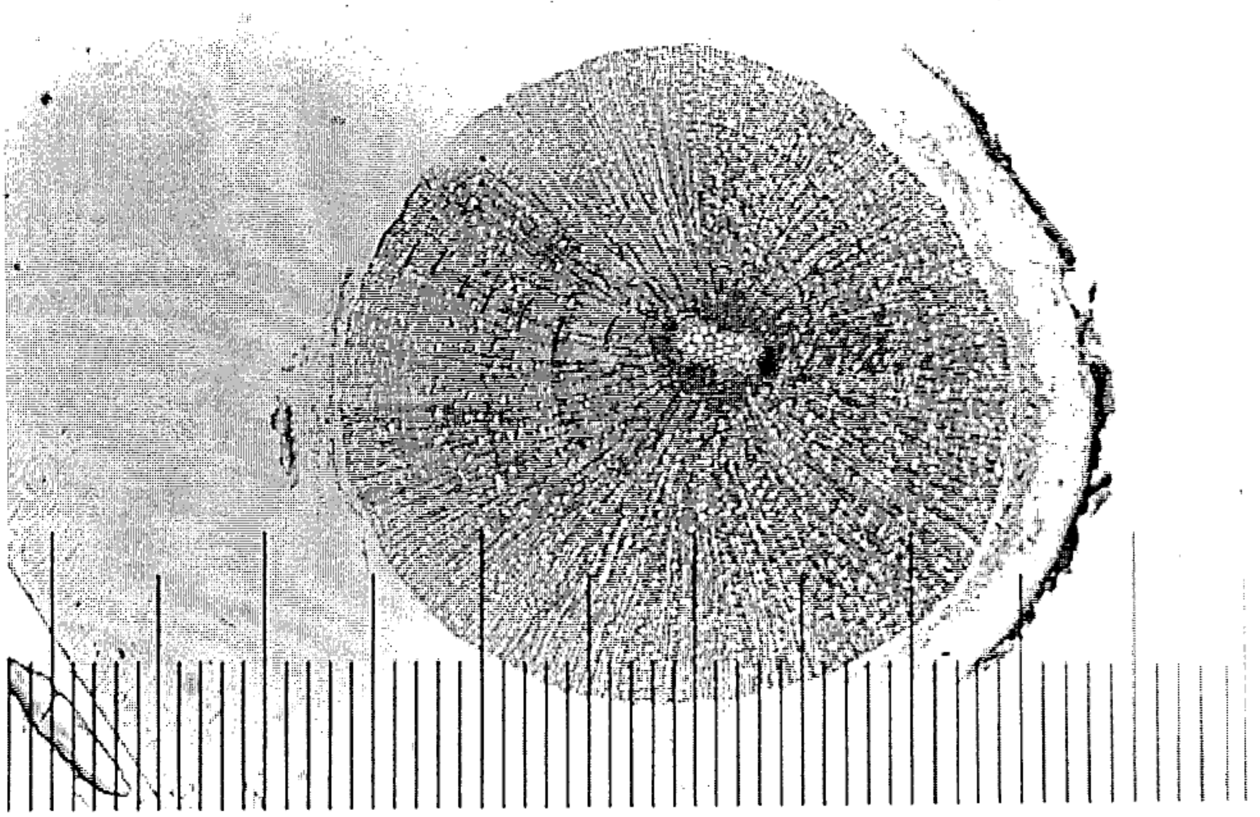


Figs 5.4-5.7 show the transverse stem sections taken from the smallest seedlings. *Nothofagus* seedlings had the clearest growth rings. *Eucryphia* and *Phyllocladus* rings were quite distinct except where there was a damaged pith (see Fig. 5.7a.) but *Atherosperma* rings were barely discernible.

The figures indicate that small seedlings of rainforest canopy species, within 20-30-year-old regeneration dominated by sclerophyllous species, are not necessarily young and that growth rates can be extremely slow. Fig. 5.5, for example, shows a 30-year-old *Eucryphia lucida* seedling which had a height of 1.4 m and a mean annual diameter increment of only 0.33 mm. The *Nothofagus* seedlings shown in Fig. 5.6 had mean annual diameter increments of 0.4 to 0.6 mm which is at least five times slower than the mean annual diameter increment of 3 mm reported by Hickey and Felton (1991) for dense 20-year-old stands dominated by *Nothofagus cunninghamii.* It appears that some rainforest seedlings in regenerating mixed forest are persisting, but barely growing, as heavily suppressed seedlings under sclerophyllous canopies.

Fig. 5.4. Transverse sections of *Atherosperma moschatum* seedlings at a 'stump height' of 0.0 m (graduations = 0.1 mm)

(a) sample 78/2, diameter = 0.3 cm, age = 8 years, diameter increment = 0.38 mm yr^{-1}



(b) sample 85/1, diameter = 0.5 cm, age = 10 years, diameter increment = 0.50 mm yr^{-1}

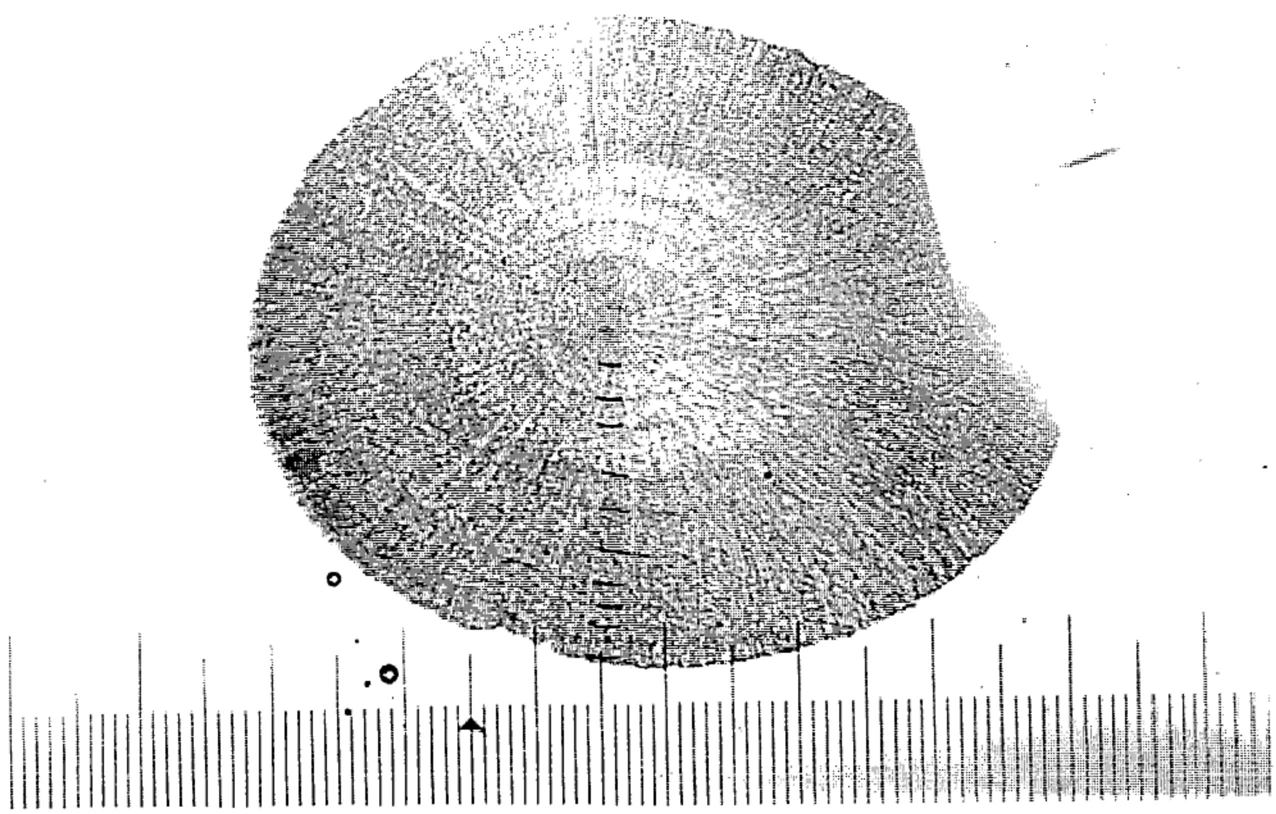


Fig. 5.4 (cont'd). Transverse sections of *Atherosperma moschatum* seedlings at a 'stump height' of 0.0 m (graduations = 0.1 mm)

(c) sample 97/2, diameter = 0.4 cm, age = 6 years, diameter increment = 0.67 mm yr $^{-1}$

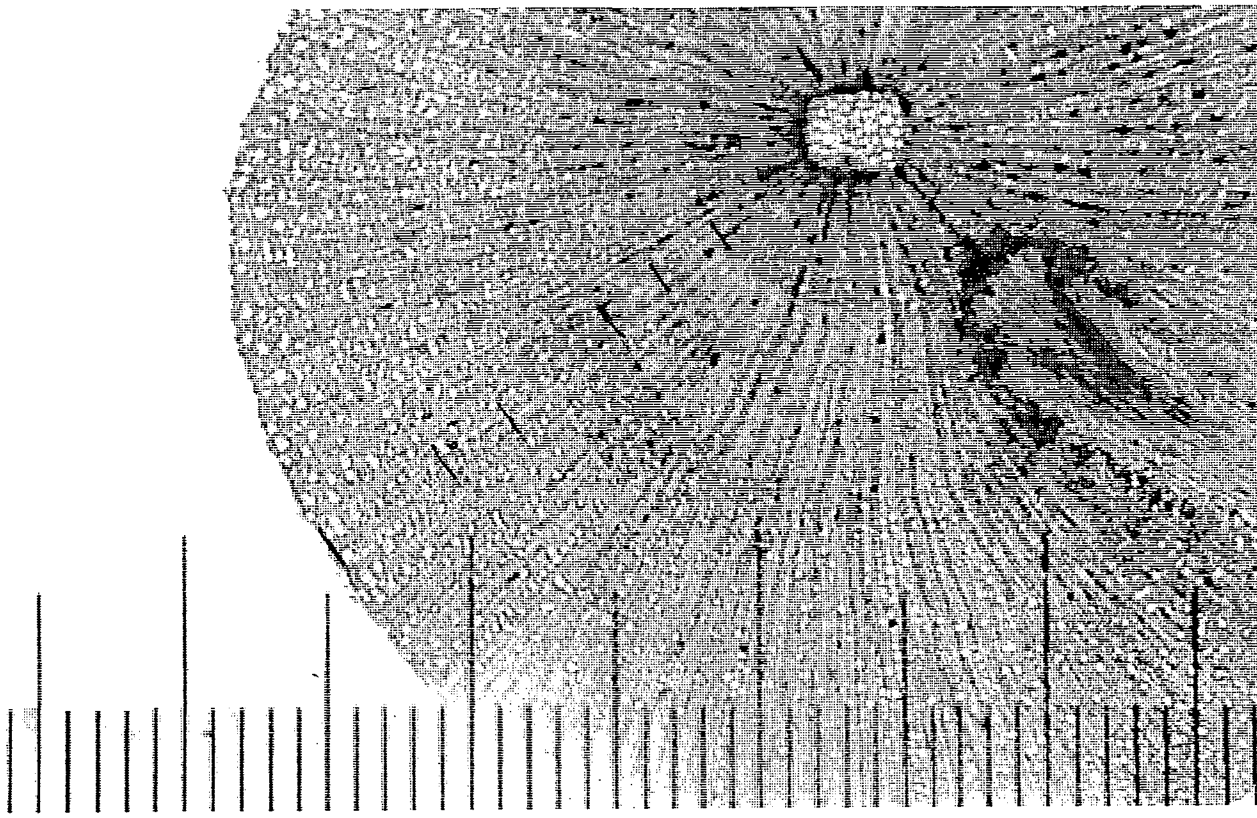


Fig. 5.5. Transverse section of a *Eucryphia lucida* seedling at a 'stump height' of 0.0 m (graduations =0.1mm)

sample 36/2, diameter = 1.0 cm, age = 30 years, diameter increment = 0.33 mm yr^{-1}

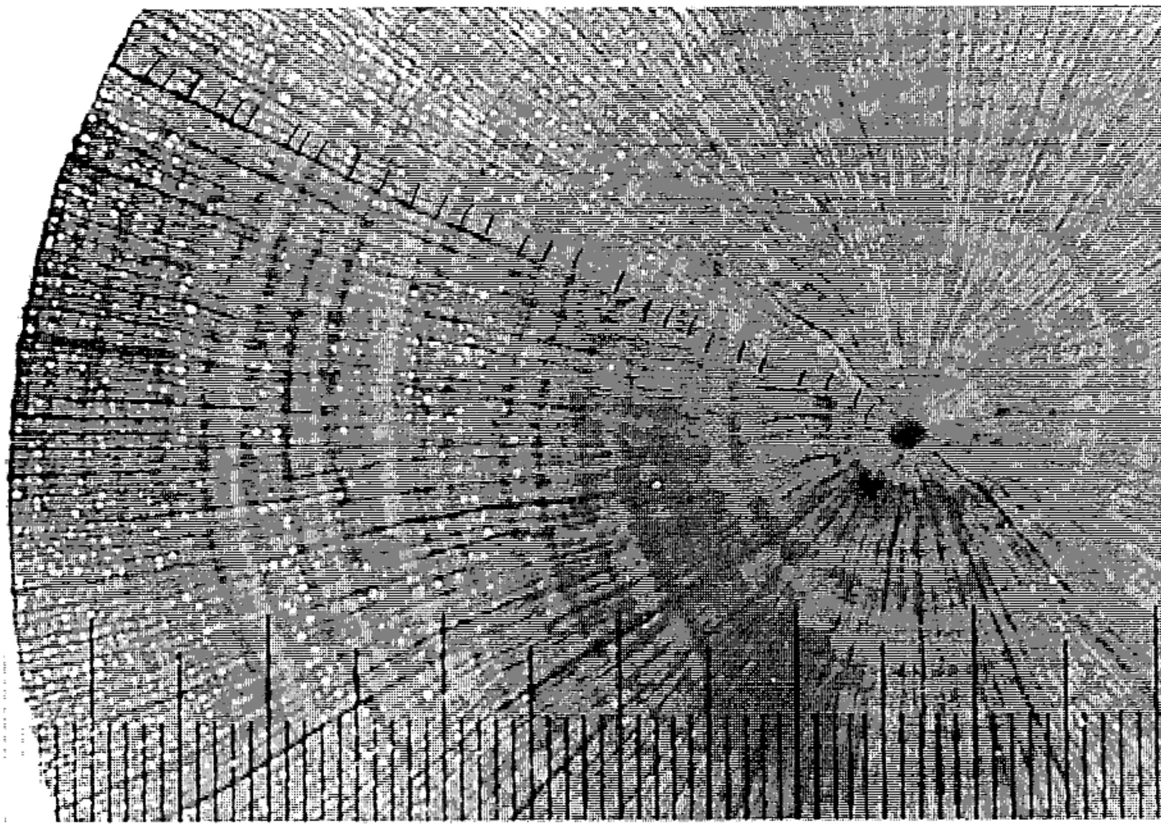
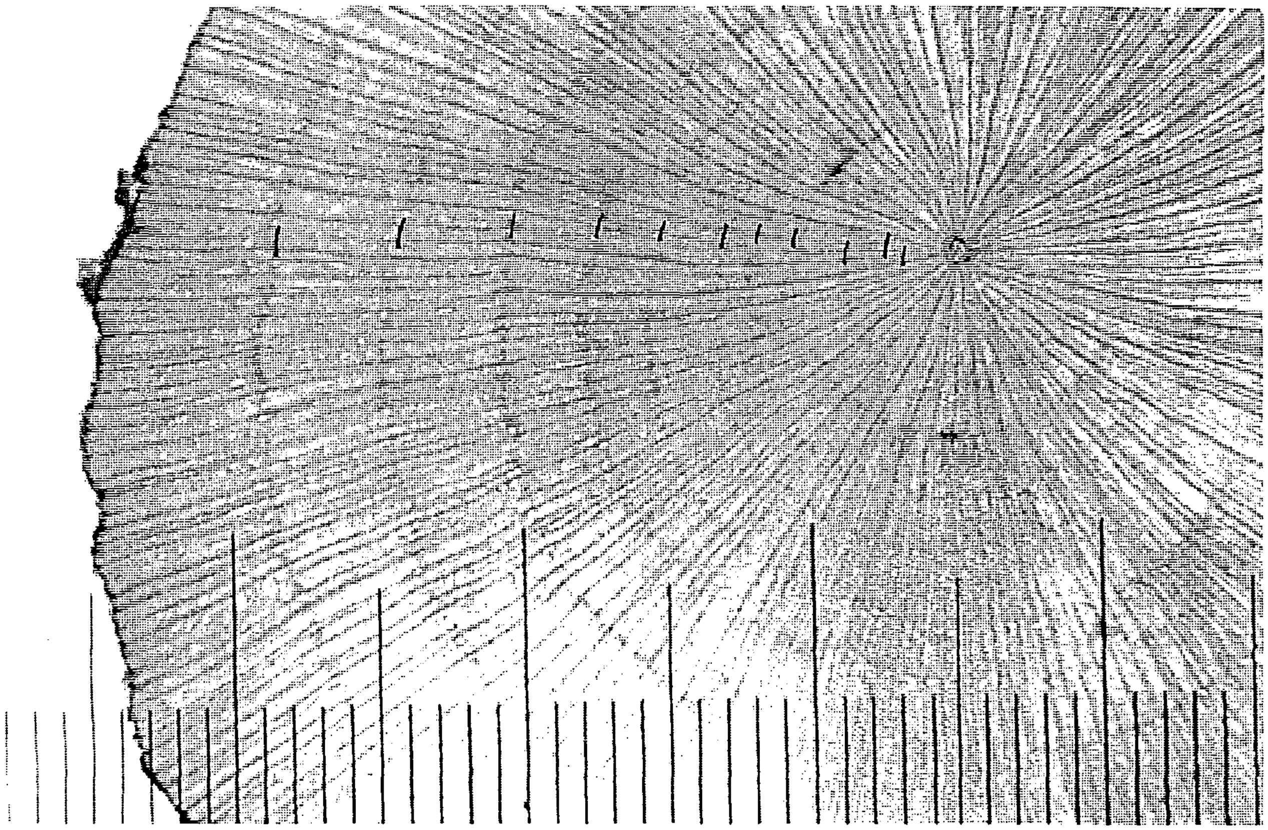


Fig. 5.6. Transverse sections of *Nothofagus cunninghamii* seedlings at a 'stump height' of 0.0 m (graduations = 0.1 mm)

(a) sample 79/2, diameter = 0.5 cm, age = 12 years, diameter increment = 0.41 mm yr^{-1}



(b) sample 89/1, diameter = 1.4 cm, age = 22 years, diameter increment = 0.63 mm yr^{-1}

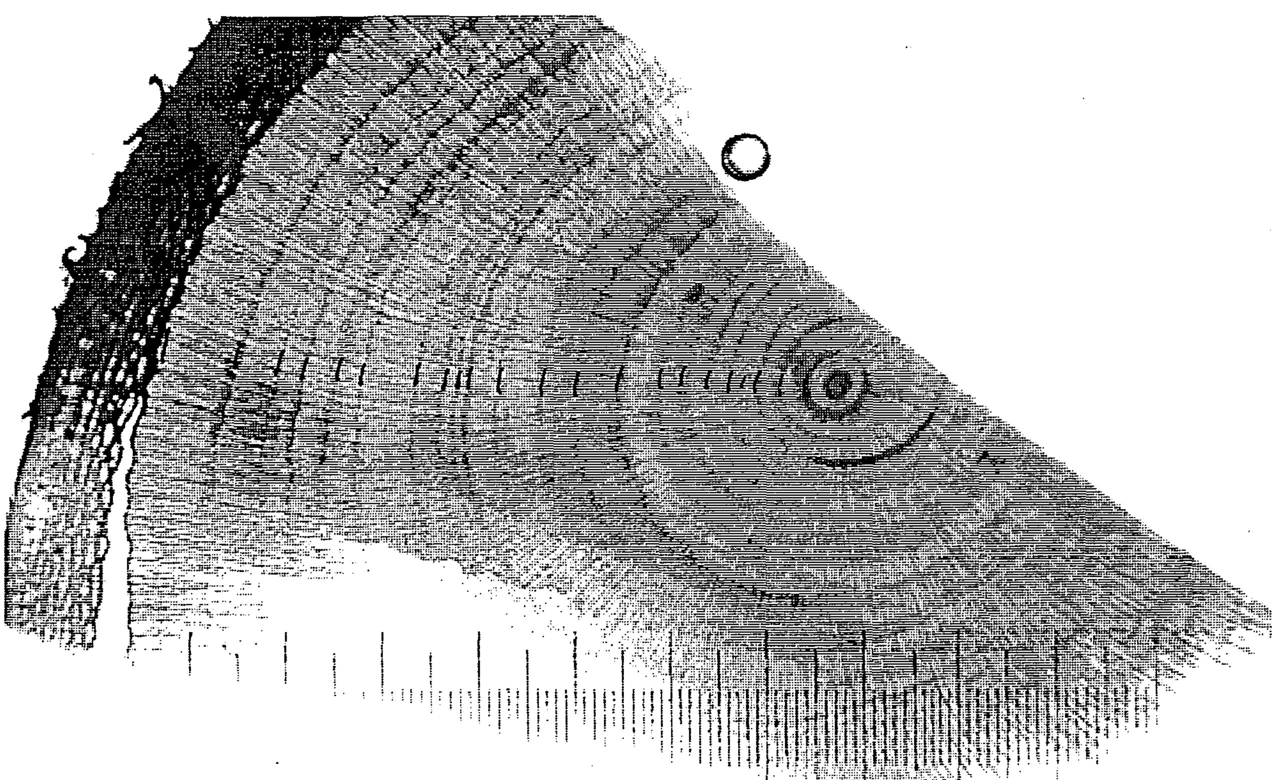


Fig. 5.6(cont'd). Transverse sections of *Nothofagus cunninghamii* seedlings at a 'stump height' of 0.0 m (graduations = 0.1 mm)

(c) sample 97/2, diameter = 0.8 cm, age = 14 years, diameter increment = 0.57 mm yr^{-1}

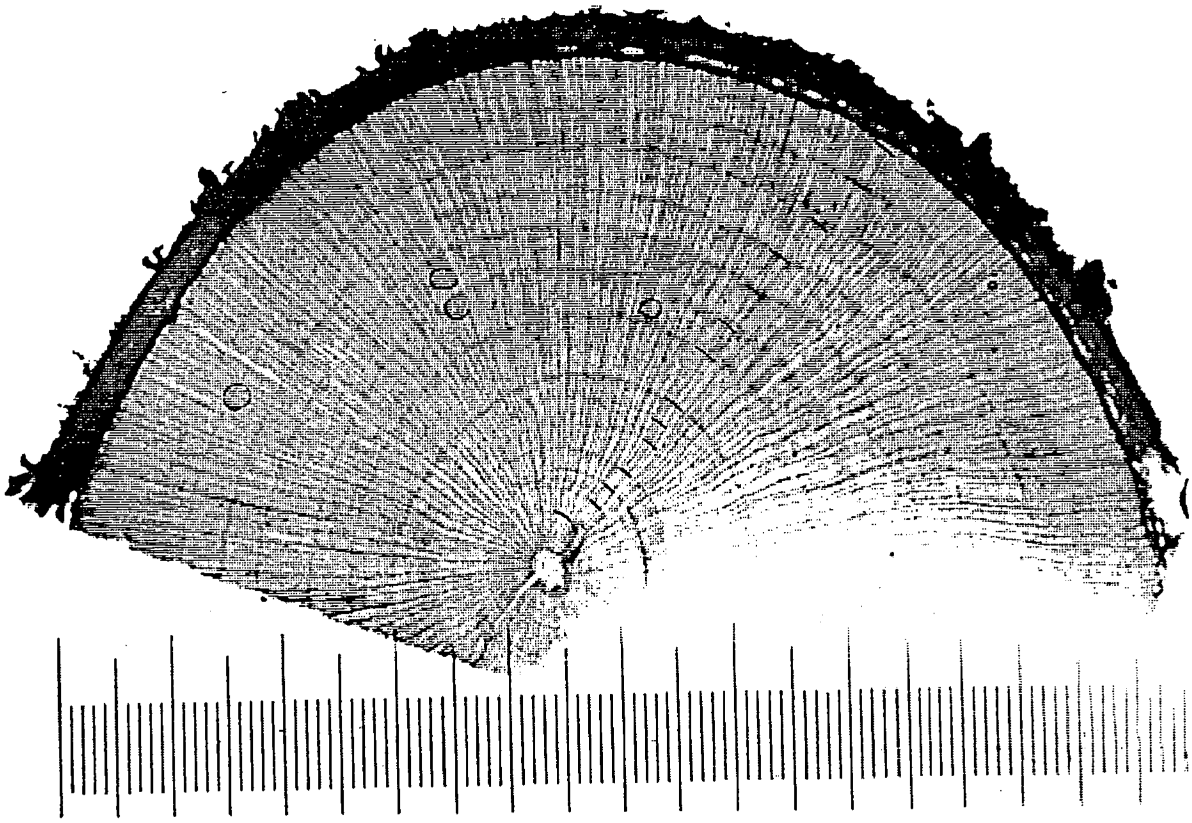
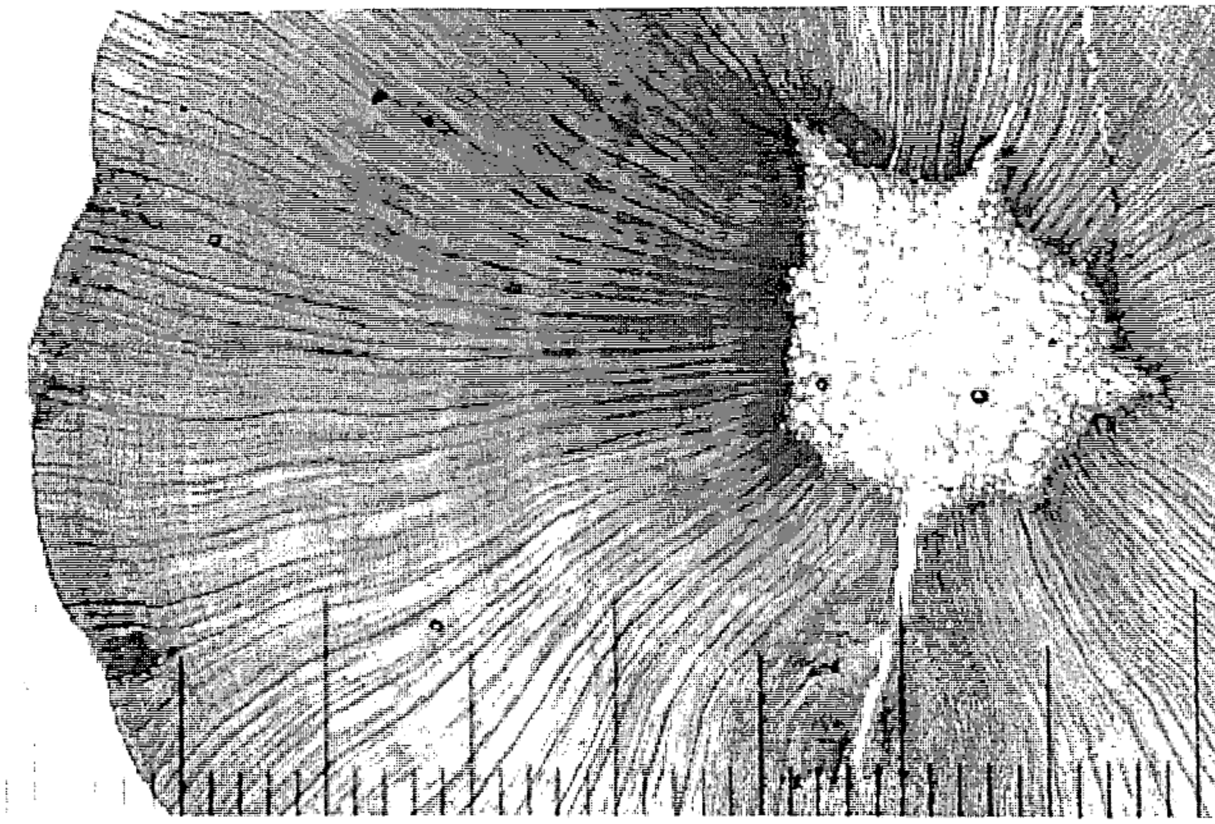
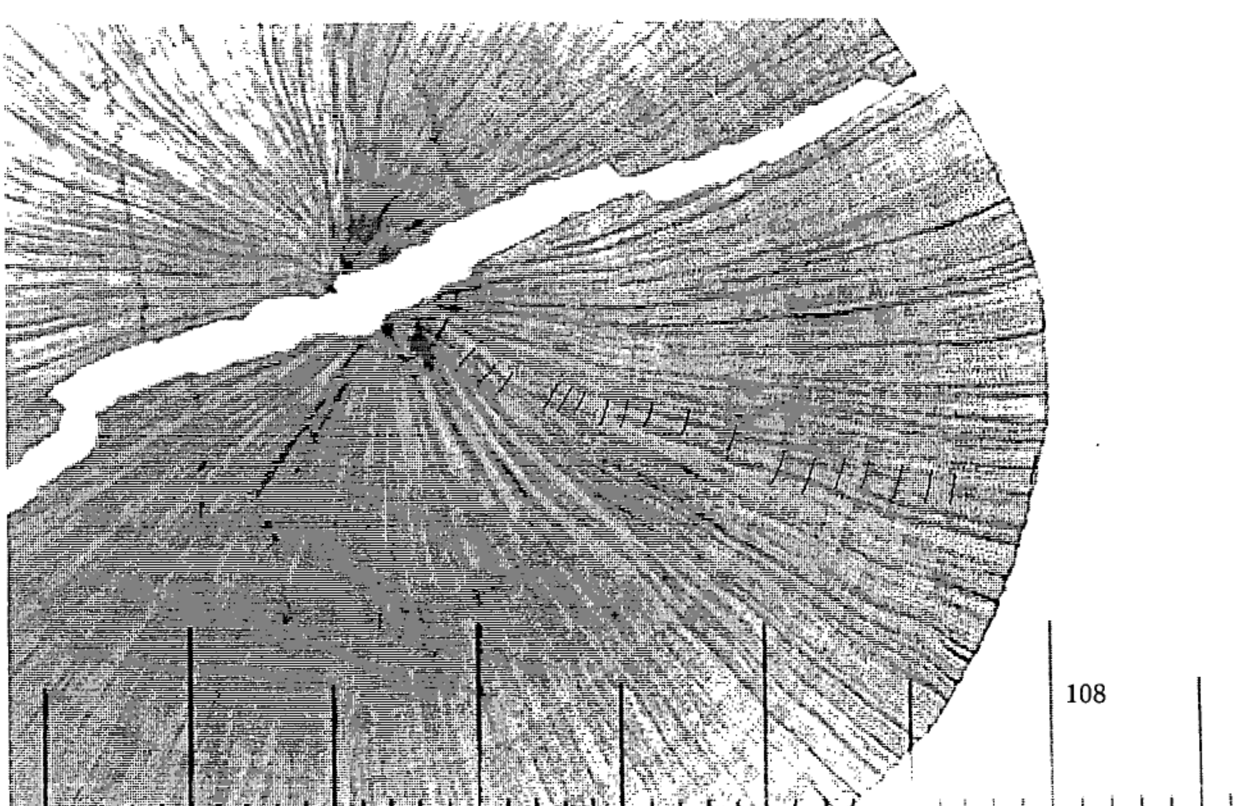


Fig. 5.7. Transverse sections of *Phyllocladus aspleniifolius* seedlings at a 'stump height' of 0.0 m (graduations = 0.1 mm)

(a) sample 62/1, diameter = 0.6 cm, pith badly rotten, indeterminate age.



(b) sample 75/2, diameter = 0.6 cm, age = 20 years, diameter increment = 0.30 mm yr^{-1}



Coupe sizes

Table 5.8 summarises information on coupe sizes in Geeveston and Smithton Districts.

Table 5.8. Number of coupes and median coupe size by district and decade.

	1960-1969	1970-1979	1980-1989	1990-1993	Total (median)
Geeveston					
No. of coupes	52	71	79	27	229
Median area (ha)	58	46	57	46	(49)
Smithton					
No. of coupes	37	43	56	13	149
Median area (ha)	64	94	174	86	(97)

The Kruskal-Wallis ANOVAs showed no significant difference in median coupe size between decades within either Geeveston or Smithton. This indicates that coupe sizes have been about the same size over the last 33 years. However, the difference in median coupe size between districts was significant ($p<0.001$). Coupes in Smithton district are about twice the size of those in Geeveston district.

If rainforest re-establishment was dependent solely on propagules from adjacent seed sources this difference in coupe size would be significant. *Nothofagus cunninghamii* has the poorest seed dispersal of the rainforest canopy trees in mixed forest. Hickey *et al.* (1982) found that the majority of *N. cunninghamii* seed fell within 20 m of a seed source and only light, non viable seed was wind-blown over a distance of 40 m. The time taken for *N. cunninghamii* seedlings to reach sexual maturity is uncertain, particularly under a dense sclerophyllous canopy, but some individuals in gaps in 20-30-year-old regeneration were observed either seeding in the present study. A simple calculation can be made of the time required for full colonisation of clearfelled coupes solely from adjacent seed sources, based on the following assumptions:

- coupes are circular and contain rainforest species in adjacent forest, at least in streamside reserves;
- 50 years is the average time for *N. cunninghamii* seedlings to reach sexual maturity in regeneration coupes; and,
- average seeding distance in regeneration coupes is 20 m.

On this basis it would take about 1000 years to fully colonise an average Geeveston coupe of 50 ha and about 1400 years for an average 100 ha coupe in Smithton district. Coupe sizes are much too large for *N. cunninghamii* seed sources from adjacent stands to fully regenerate the logged area within the planned rotation age. However, this study and those of Jordan *et al.* (1992), Felton and Lockett (1983) and Calais, reported in Hickey and Savva (1992), have shown that regeneration of rainforest canopy species, including *N. cunninghamii,* is quite common within logged coupes and not dependent on adjacent seed sources. Thus the time required for *N. cunninghamii* to fully recolonise a coupe is much less than the times indicated. The frequently observed regeneration of this species within coupes must occur from vegetative sources, seed or seedlings which have survived the regeneration burn, or seed shed subsequently by surviving mature trees.

Conclusion

The information presented in this chapter can be combined with previous studies to summarise the reported regeneration mechanisms for common rainforest species in Tasmanian lowland mixed forests. Table 5.9 lists the ~~seed~~ fruit morphology and regeneration mechanisms of rainforest angiosperm and gymnosperm species which had a mean frequency of at least 5 percent in the oldgrowth mixed forest sites described in Chapter 3.

The table indicates that most, and possibly, all rainforest angiosperms or gymnosperms which are common in mixed forest have some ~~strategy~~ characteristic which enables them to recolonise sites after massive disturbance. Species with short seed dispersal distances such as *Nothofagus cunninghamii, Eucryphia lucida* and *Anodopetalum biglandulosum,* all have some ability to sprout after fire. Species which do not sprout after fire mostly have a capacity for long distance dispersal of their seeds by either wind or birds. Almost half the species can also regenerate from seed stored in the soil.

Much less is known of the regeneration mechanisms of fern species but all are capable of regeneration from spores that can be transported by wind, water or on animal bodies. Howard (1974) recorded regeneration of *Histiopteris incisa* and *Hypolepis incisa* from soil samples which suggests that spores for these species are stored in the soil. Cremer and Mount (1965) noted the ability of *Dicksonia antarctica* to resprout after fire. All the rainforest ground ferns appeared to recover well after wildfires or clearfelling and burning in the present study (see Chapter 3), but it is uncertain what proportion of the regeneration was due to resprouting from rhizomes rather than from spores. Some epiphytic ferns, such as *Hymenophyllum* spp., appear to require protected moist microsites which are uncommon in young forests.

Table 5.9. Fruit morphology and regeneration mechanisms for common angiosperms and gymnosperms in oldgrowth mixed forest.

Species	Fruit type[5, 15]	Soil seed bank[10,12]	Primary dispersal agent	Max. seed dispersal distance	Sprouts after fire
Trees				m	
Atherosperma moschatum	achene	no	wind[7]	>200[7]	yes (weakly)[3,4,12]
Eucryphia lucida	leathery capsule	no	wind[7]	<200[7]	yes (weakly)[3,8,12]
Nothofagus cunninghamii	woody nutlets	no	wind[7]	<200[7]	yes (weakly)[3,9,12]
Phyllocladus aspleniifolius	aril	yes[12,13]	birds[11,13]	>200	no[3,12]
Tall shrubs					
Anodopetalum biglandulosum	fleshy	no	wind[11]	<200	yes (weakly)[3,8]
Anopterus glandulosus	capsule	no	?	?	yes (weakly)[3,8]
Cenarrhenes nitida	drupe	no	birds[11]	>200	yes (weakly)[3]
Olearia argophylla	achene	no	wind[4]	>200	yes[2,4]
Monotoca glauca	berry	yes[12]	birds[14]	>200	no[3]
Pittosporum bicolor	capsule	yes[10]	birds[14]	>200	no[3]
Tasmannia lanceolata	berry	yes[10]	birds[14]	>200	no[3]
Short shrubs					
Aristotelia peduncularis	berry	no	?birds[1]	>200	no[11]
Coprosma quadrifida	drupe	yes[10]	birds[6,14]	>200	no[3]
Cyathodes juniperina	drupe	yes[12]	birds[14]	>200	no[11]
Pimelea drupacea	drupe	no	birds?	?	no[11]
Trochocarpa cunninghamii	drupe	yes[12]	birds?	?	yes (weakly)[3]
Herbs and sedges					
Dianella tasmanica	berry	no	birds?[6]		no[11]
Gahnia grandis	nut	yes[12]	birds[6]	>200	?
Hydrocotyle spp.	schizocarp	yes[10,12]	?	?	no[3]
Climbers					
Clematis aristata	achene	no	wind	>200m	?
Prionotes cerinthoides	capsule	?	?	?	?

1. Armesto and Rozzi (1989) noted bird dispersal of *Aristotelia chilensis* (Mol.) Stuntz
2. Ashton (1981[b])
3. Barker (1991)
4. Cremer and Mount (1965)

5. Curtis (1963), Curtis (1967), Curtis and Morris (1975)
6. French (1990)
7. Hickey *et al.* (1982)
8. Hill and Read (1984)
9. Howard (1973)
10. Howard (1974)
11. Personal observation
12. Present study
13. Read (1989)
14. Read and Hill (1983)
15. Willis (1970)

6. DENSITY AND GROWTH OF SPECIAL TIMBER SPECIES IN REGROWTH EUCALYPT FORESTS

In Tasmania the term 'regrowth eucalypt forest' is frequently applied to wet eucalypt forests which are less than 110 years old. As well as silvicultural regeneration, regrowth forest also include stands that have resulted from wildfires and/or logging prior to the introduction of routine silvicultural regeneration techniques in about 1960. As the origin of these stands generally pre-dates the existence of PI maps or aerial photography it is often not possible to estimate the nature of the pre-regrowth forest although it is possible to determine its height potential from occasional oldgrowth remnants or stags. In areas of higher rainfall it is reasonable to assume that much of the regrowth forest with a tall height potential would previously have been oldgrowth mixed forest.

This chapter includes an analysis of data from permanent plots in stands established since 1960 where the origin of the stands and the nature of the pre-regrowth forests is known from PI maps. This analysis allowed a comparison of the occurrence of eucalypt and special timber species in regeneration which has resulted from a wildfire, retained eucalypt seed trees after logging, or following clearfelling, slash burning and artificial sowing of seed. The major special timber species in mixed forest include blackwood (*Acacia melanoxylon*), celery-top pine (*Phyllocladus aspleniifolius*), leatherwood (*Eucryphia lucida*), myrtle (*Nothofagus cunninghamii*), sassafras (*Atherosperma moschatum*) and silver wattle (*Acacia dealbata*).

This chapter also describes the frequency, diameter growth and density of eucalypt and special timber species in older wet eucalypt regrowth forests of the Geeveston district. The approximate rotation length required for special timber species to reach commercial sawlog sizes is estimated.

Method

Occurrence of special timbers in regrowth stands regenerated since 1960

The Tasmanian Forestry Commission has a Continuous Forest Inventory (CFI) program which relies on remeasurement of permanent plots in order to monitor tree growth on State forests. One type of permanent plot, known as a CFI-Regen plot (Forestry Commission 1985), is established on a stratified random basis in regeneration areas at a rate of one 0.1 ha plot per 250 ha. A total of about 140 of these plots have been established in silvicultural regeneration, and occasionally in regeneration resulting from wildfires, once the forest has reached an age of about 10 years. This program extends over most State forests although plots do not exist for some areas with mixed forest, e.g. the West Coast District. The plots are primarily designed to provide wood volume information. All commercial tree species with diameters greater than 10 cm at breast height are recorded and the plots are remeasured at least once every 10 years. Since the early 1980s the measurements have included an estimate of the density of rainforest tree species which are less than 10 cm diameter at breast height (dbh). As part of the current project a study was made of the occurrence of special timbers species regeneration on CFI-Regen plots. The previous PI types for the plots were determined by reference to archived PI maps so that the regeneration could be compared with the former forest type. Thirty-six of the plots were located in coupes which were previously mixed forest with a mature *Nothofagus cunninghamii* understorey and 22 plots were previously rainforest with mature *N. cunninghamii* in the canopy. Thirty-two of the mixed forest plots had a mature eucalypt height of at least 41m while four plots were located on sites with a eucalypt height potential of 27-41m. The plots are classed according to their regeneration type:

- W - regeneration which results from a wildfire;
- N - regeneration which results from natural eucalypt seedfall, usually from seed trees retained from logging; or
- A - regeneration which results from artificial sowing of eucalypt seed.

The wildfire regeneration has usually been salvage logged after the fire which has probably affected the regeneration.

The frequency of eucalypt and special timbers species was determined for each plot type. The PI maps of the previous forest indicated that all plots formerly contained mature *Nothofagus cunninghamii.* A Chi-squared analysis was used to compare the frequency of *N. cunninghamii* in the regenerated stands to determine if there were significant differences in frequency according to previous forest type or regeneration type.

Frequency, diameter growth and density of special timbers in regrowth stands regenerated prior to 1960

The Forestry Commission maintains permanent plots, referred to as CFI-Regrowth plots, in eucalypt regrowth forests established without silvicultural treatment. These 0.2 ha rectangular plots are randomly allocated to stratified forest classes at a rate of one per 250 ha, and are initially measured five years after establishment and then at 10 year intervals (Forestry Commission 1985). For the purposes of this project the data from CFI-Regrowth plots with a potential eucalypt height of at least 41m in Geeveston District were examined to determine the average density and diameter of eucalypt and special timber species. A map of the occurrence of eucalypt forests with a potential height of at least 41m (Forestry Commission 1988) was overlain with a map of land systems (Davies 1988). This indicated that most of these forest areas in Geeveston District have an annual rainfall of at least 1000 mm and it was concluded that a substantial proportion of the Geeveston regrowth sites would previously have supported mixed forest.

A list of CFI-Regrowth plots in the Geeveston District was obtained from a computerised database. The frequency, mean age, diameter and density of commercial species greater than 10 cm dbh was obtained from the most recent measurement of CFI plots which were PI-typed as being pure regrowth. Plots which occurred in stands which were mapped as being a mixture of oldgrowth and regrowth trees were excluded from this analysis. Plots which had more than one age of regrowth were also excluded as this implied that these stands had been subject to multiple disturbances, particularly fire, and were less likely to contain rainforest tree species than single-aged regrowth. Fifty-nine plots, with a eucalypt height potential of at least 41m, were used to estimate:

- the frequency and density of eucalypt and special timber species in pure regrowth stands;
- the correlation between stand age, mean diameter and density for eucalypt and special timber species; and

- the linear relationship between age and diameter for eucalypt and special timber species.

A comparison of local mean and current annual diameter increments of eucalypt and special timbers species was made for two stands with two plot measurements. The scope for this work is currently limited because information on special timbers species other than silver wattle has only been collected since 1979 and few plots with a high component of special timbers species have been measured twice since that time. For example, no CFI plots in Geeveston District have yet had successive measurements of *Phyllocladus aspleniifolius*. Although this species was recorded in three plots, in each case the trees had only just attained the minimum diameter limit for measurement of 10 cm dbh. Plot 1075, established in 1986 and remeasured in 1990, contained all special timbers species except *Acacia dealbata* and *Phyllocladus aspleniifolius* and was selected to obtain an accurate local estimate of mean and current annual diameter increment for special timbers species in tall mixed forest regrowth. The stand originated in 1922, hence the annual growth estimates are calculated over the period from age 64 to 68 years. The plot was PI-typed as ER2/1&2 from 1947 photography and ER4c/1&2 from 1984 photography. The data extracted from the plot included PI type, species, diameter of all commercial tree species greater than 10 cm dbh, and canopy class. Canopy class divides trees as follows: 1 = dominant; 2 = co-dominant; 3 = subdominant and 4 = suppressed. Only trees which were alive at both measurements were included in the analysis. Data from plot 1046, established in 1967 and remeasured in 1972, 1980 and 1990, were examined to obtain local mean and current annual diameter increment estimates for *Acacia dealbata*. This stand also originated in 1922 and was PI-typed as ER2/2 from 1947 photography and ER4b/2 from 1984 photography. A number of other CFI plot sheets with special timbers were examined but the presence of large diameter rainforest species indicated that these stands included survivors from the previous forest and therefore were unsuitable for comparisons of growth rates between species.

Results

Analysis of CFI-Regen plots

Table 6.1 shows the frequency of plots with eucalypt and special timbers species according to previous forest type and regeneration type. The table shows that 57 percent of mixed forest and rainforest plots contained *Nothofagus cunninghamii* in subsequent regeneration at about 20 years of age.

Table 6.1 Occurrence of commercial tree species on CFI plots in regeneration coupes which were previously mature rainforest or mixed forest. (W = regenerated from a wildfire; N = naturally seeded with eucalypts; A = artificially sown with eucalypts) (N. cun. = *Nothofagus cunninghamii*, A. mel. = *Acacia melanoxylon*, E. luc. = *Eucryphia lucida*, A. mos. = *Atherosperma moschatum*, P. asp. = *Phyllocladus aspleniifolius*, A. dea. = *Acacia dealbata*, Euc. = *Eucalyptus* spp.)

Previous forest	Regen. type	Plots	Mean age (years)	Frequency						
				N. cun.	A. mel.	E. luc.	A. mos.	P. asp.	A. dea.	Euc.
Rainforest	W	6	21	3	1	1	2	1	0	6
				50%	17%	17%	33%	17%	0%	100%
	N	1	21	0	0	0	0	0	1	0
				0%	0%	0%	0%	0%	100%	0%
	A	15	17	10	4	4	4	5	3	15
				67%	27%	27%	27%	33%	20%	100%
	TOTAL	22	18	13	5	5	6	6	4	21
				59%	23%	23%	27%	27%	18%	95%
Mixed	W	10	22	6	3	1	3	3	4	10
forest				60%	30%	10%	30%	30%	40%	100%
	N	5	22	3	3	2	1	3	1	5
				60%	60%	40%	20%	60%	20%	100%
	A	21	19	11	8	4	5	6	1	21
				52%	38%	19%	24%	29%	5%	100%
	TOTAL	36	20	20	14	7	9	12	6	36
				56%	39%	19%	25%	33%	17%	100%
GRAND		58	19	33	19	12	15	18	10	57
TOTAL				57%	33%	21%	26%	31%	17%	98%

The absence of *Nothofagus cunninghamii* regeneration ~~for~~ in some plots could be attributed partly to the occurrence of wildfires or regeneration burns in years when there was little *N. cunninghamii* seed present. Hickey *et al.* (1982) found that *N. cunninghamii* seeds heavily every two or three years but there is very little seed produced in non-mast years. In non-mast years *N. cunninghamii* regeneration would be mostly reliant on vegetative regeneration. While the PI types for all the plots indicated that *N. cunninghamii* was present in the previous forest there is no information on the original occurrence of other special timbers species. At least one fifth of the plots contained *Atherosperma moschatum, Eucryphia lucida* or *Phyllocladus aspleniifolius* in the regeneration. *Acacia melanoxylon* and *A. dealbata* occurred in 33 and 17 percent respectively of the plots, which probably represents an increased frequency of these species compared with the previous forest.

Table 6.2 summarises the number of plots, according to previous forest type, which contained *Nothofagus cunninghamii* in the regenerated stand. The p-value for the Chi-squared analysis was 0.71 and indicated that there was no significant association between former forest type and the presence of *N. cunninghamii* in the regeneration.

Table 6.2. Number of plots with *Nothofagus cunninghamii* arranged by previous forest type.

Previous forest	*Nothofagus cunninghamii*		
	present	absent	proportion present
Rainforest	13	9	0.59
Mixed forest	20	16	0.56

The data for each forest type were combined to observe the association, if any, between regeneration type and the presence of *N. cunninghamii* in the regeneration. Table 6.3 summarises the number of plots, according to regeneration type, which contained *N. cunninghamii* in the regenerated stand.

Table 6.3. Number of plots, according to regeneration type, with *Nothofagus cunninghamii* .

Regeneration type	*Nothofagus cunninghamii*		
	present	absent	proportion present
W	9	7	0.69
N	3	6	0.50
A	21	15	0.58

The p-value for the Chi-squared test of association was 0.93 and thereby indicated that there was no significant difference in the success of *N. cunninghamii* regeneration among the three regeneration types. Even after the most severe treatment, i.e. clearfelling, burning and artificial sowing with eucalypts, 58 percent of the plots retained

N. cunninghamii on the immediate site. The rainforest sites burnt by a wildfire appear to have contained, or have been adjacent to, eucalypt trees as they are all stocked with eucalypts.

Analysis of CFI-Regrowth plots.

Table 6.4 shows the frequency, mean age, diameter, annual diameter increment and stem density of eucalypt and special timbers species in pure regrowth stands. *Eucalyptus obliqua* was the most frequent tree species and occurred on nearly three quarters of the regrowth plots. *Acacia dealbata* and *A. melanoxylon* were the most abundant special timbers species and were recorded on about one third of the plots. *Nothofagus cunninghamii* and *Atherosperma moschatum* were also common and were recorded on at least one fifth of the plots. *Eucryphia lucida* and *Phyllocladus aspleniifolius* were only occasionally recorded. Special timbers species, where present, had mean stem densities of at least 50 stems ha^{-1}. This may be a marked underestimate of the true frequency and density of the rainforest species, particularly for *P. aspleniifolius*, as their slow growth rates may have prevented many individuals from reaching the required minimum diameter of 10 cm dbh in order to be recorded.

Table 6.4. Frequency, mean stand age, mean diameter, mean diameter increment and density of eucalypt and special timber species in pure regrowth stands in Geeveston District.

Species	Frequency (%)	Mean age (years)	Mean diameter (cm)	Mean dbh inc. (cm yr^{-1})	Mean density (stems ha^{-1})
Acacia dealbata	34	64	26.4	0.41	66
Acacia melanoxylon	36	61	24.8	0.41	50
Atherosperma moschatum	20	67	14.1	0.21	68
Eucryphia lucida	5	71	18.5	0.26	206
Eucalyptus delegatensis	5	64	27.5	0.43	295
Eucalyptus globulus	12	65	49.2	0.76	152
Eucalyptus johnstonii	2	65	16.9	0.26	290
Eucalyptus obliqua	73	64	40.1	0.63	408
Eucalyptus regnans	25	59	41.8	0.71	100
Nothofagus cunninghamii	24	60	16.7	0.28	164
Phyllocladus aspleniifolius	3	71	11.4	0.16	110

Table 6.5 shows the correlation between diameter and age, and between diameter and eucalypt density, for eucalypt and special timbers species which were recorded on at

least ten CFI-Regrowth plots. Values of the R^2 statistic indicate the percentage of the variation in diameter which can be explained by age.

Table 6.5. Correlation between diameter (dbh) and age, diameter and eucalypt density, and R^2 values for the linear relationship between diameter and age for eucalypt and special timbers species in pure regrowth stands.

Species	N	Correlation of dbh with:		R^2
	(obs)	age	eucalypt density	%
Acacia dealbata	15	0.00	-0.15	2
Acacia melanoxylon	17	-0.19	-0.43	2
Atherosperma moschatum	10	-0.75	-0.68	35
Eucalyptus obliqua	43	0.23	-0.61	5
Eucalyptus regnans	15	0.50	-0.31	25
Nothofagus cunninghamii	11	0.43	-0.65	14

Table 6.5 indicates there is very poor correlation between mean diameter and age for special timbers species when mean plot growth rates are compared. Only *Nothofagus cunninghamii* had a positive correlation between diameter and age while *Acacia melanoxylon* and *Atherosperma moschatum* had negative correlations. These implausible results may be due to factors such as:

- large variation in growth rates between plots;
- periodic, or continuous, regeneration of special timbers species after the establishment of the regrowth eucalypts; and
- the occurrence of trees which are older than the eucalypt regrowth and which are remnants of the pre-existing forest.

These data do not allow useful predictions of special timbers growth. However at a gross level the mean diameter increments shown in Table 6.4 suggest that the special timbers species will be less than a desired minimum commercial diameter, e.g. of 50 cm dbh, at the planned rotation age of 90 years.

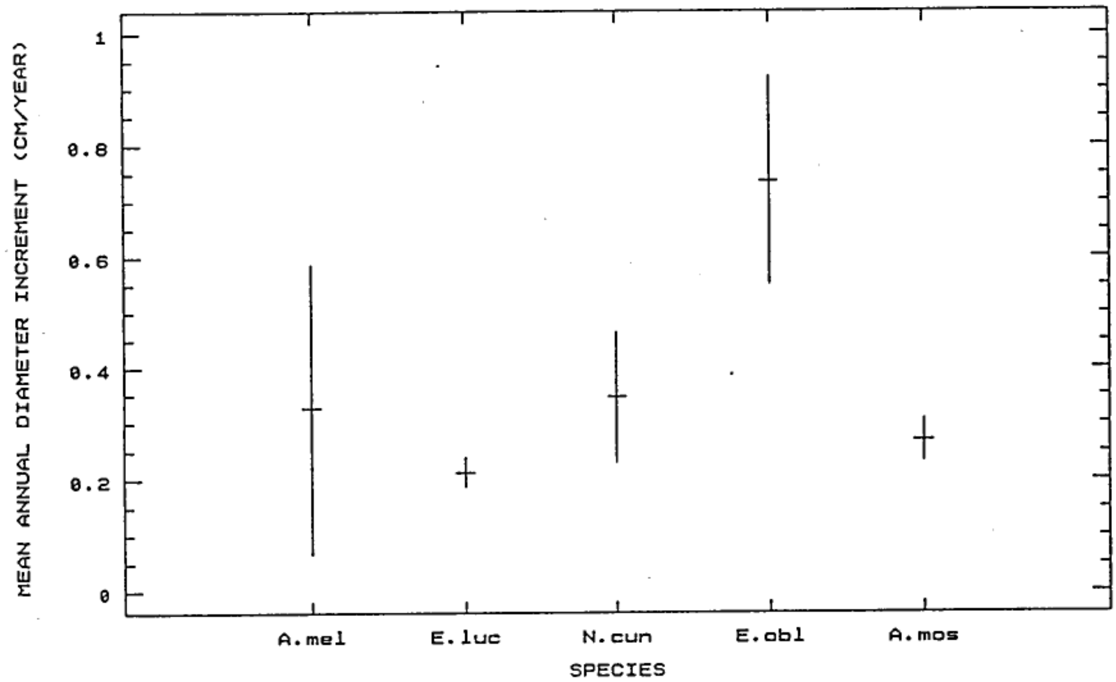
Table 6.6 summarises data from CFI plot 1075 and Fig. 6.1 shows estimates of mean and current annual diameter increment, with 95 percent confidence limits, for each species. Mean annual increments ranged from 0.2 cm yr^{-1} for *Eucryphia lucida* to 0.8 cm yr^{-1} for *Eucalyptus obliqua*. Fig. 6.1 shows that *E. obliqua* had a significantly higher mean annual diameter increment than all other species with the possible exception of *Acacia melanoxylon* which had a very imprecise mean annual diameter increment estimate due to the low sample size.

Table 6.6. Number of trees in each canopy class, mean diameter at 68 years (± standard deviation), mean and current annual diameter increment (± standard deviation) for the period from 1986 to 1990 for CFI plot 1075. (n_1 = dominant, n_2 = codominant, n_3 = subdominant, n_4 = suppressed)

Species	Trees	Canopy class n_1,n_2,n_3,n_4	Mean dbh (cm)	Mean dbh inc. (cm y^{-1})	Current dbh inc. (cm y^{-1})
Acacia melanoxylon	3	0 0 2 1	21.9±7.2	0.33	0.08±0.14
Atherosperma moschatum	16	0 0 3 13	17.7±4.8	0.27	0.13±0.06
Eucalyptus obliqua	17	5 5 2 5	48.9±23.2	0.78	0.23±0.36
Eucryphia lucida	20	0 0 1 19	13.8±3.5	0.22	0.16±0.10
Nothofagus cunninghamii	8	1 1 2 4	22.1±9.0.	0.35	0.30±0.05

Current annual diameter increments for special timbers species in 68-year-old forest were very low, i.e. from 0.08 to 0.30 cm yr^{-1}, and lower than the corresponding mean annual diameter increments which suggests the growth of trees has slowed. Table 6.6 shows that most special timber species trees were classed as either suppressed or subdominant trees. However they maintained positive, albeit very slow, growth. This contrasts with suppressed eucalypt trees which showed no increment, or a decrement, over the 5-year period. However, even if these trees are ignored, the eucalypt current annual diameter increment was only 0.34 cm yr^{-1} and only marginally faster than for the *Nothofagus cunninghamii.* The plot measurement sheet indicated that all five co-dominant, and one dominant, eucalypt trees showed signs of crown dieback which is a common condition in eucalypt regrowth in southern Tasmania (Podger *et al.* 1980). Fig. 6.1 indicates that mean current annual diameter increments of all species were not significantly different, at the 95 percent confidence level.

Fig. 6.1. Mean annual diameter increment and current annual diameter increment for a 68-year-old stand (plot 1075). (A.mel = *Acacia melanoxylon*, E.luc = *Eucryphia lucida*, N.cun = *Nothofagus cunninghamii*, E.obl = *Eucalyptus obliqua*, A.mos = *Atherosperma moschatum*)



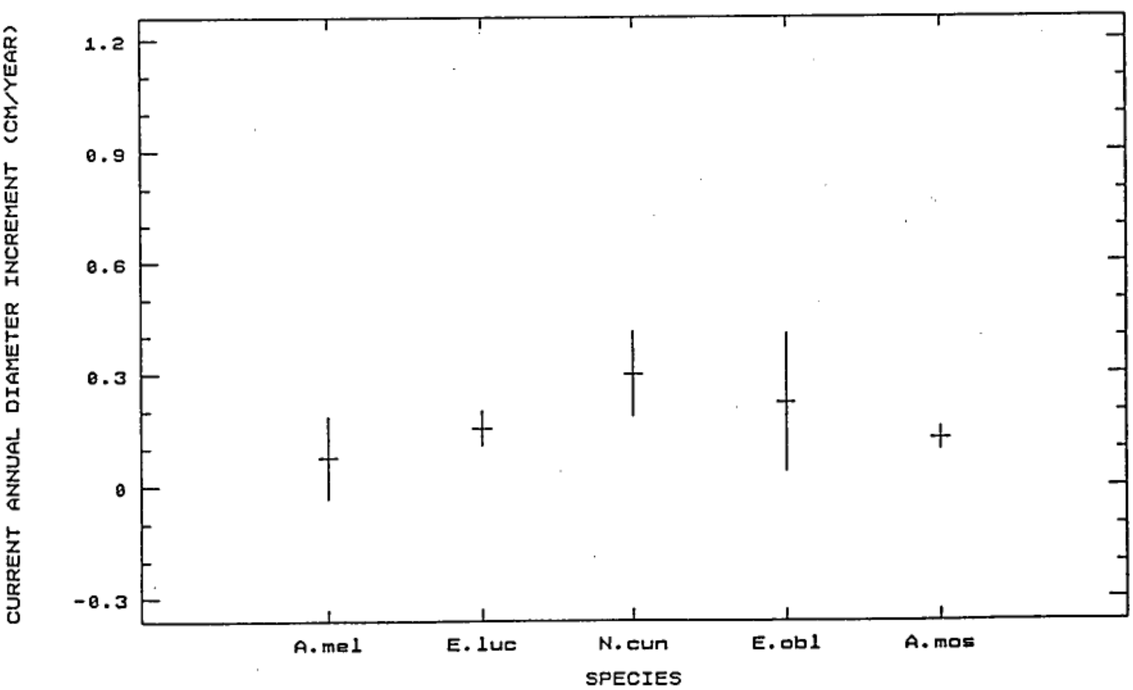


Table 6.7 shows the mean diameter for each species which would be attained by this stand if the mean annual diameter increment was maintained for a 90-year rotation. These diameters may be optimistic in view of the current annual diameter increment estimates presented in Table 6.6.

Table 6.7. Expected mean diameters after 90 years for plot 1075.

Species	Mean dbh (cm)
Acacia melanoxylon	30
Atherosperma moschatum	24
Eucalyptus obliqua	70
Eucryphia lucida	20
Nothofagus cunninghamii	32

Table 6.8 summarises data from plot 1046 for the period from 1967 to 1972. The plot contained 14 live *Acacia dealbata* trees over this period but only three trees remained alive in 1980 and only one tree was alive in 1990. This result is consistent with Gilbert (1959) who observed that *Acacia dealbata* trees rarely live for more than 70 years. One individual each of *Atherosperma moschatum* (dbh 14.2 cm) and *Nothofagus cunninghamii* (dbh 19.8 cm) were recorded at the 1990 measurement and thereby ~~indicated~~ that the stand was developing mixed forest.
suggested

Table 6.8. Number of trees in each canopy class, mean diameter at 50 years (± standard deviation), mean and current annual diameter increment (± standard deviation) for the period from 1967 to 1972 for CFI plot 1046. (n_1 = dominant, n_2 = codominant, n_3 = subdominant, n_4 = suppressed)

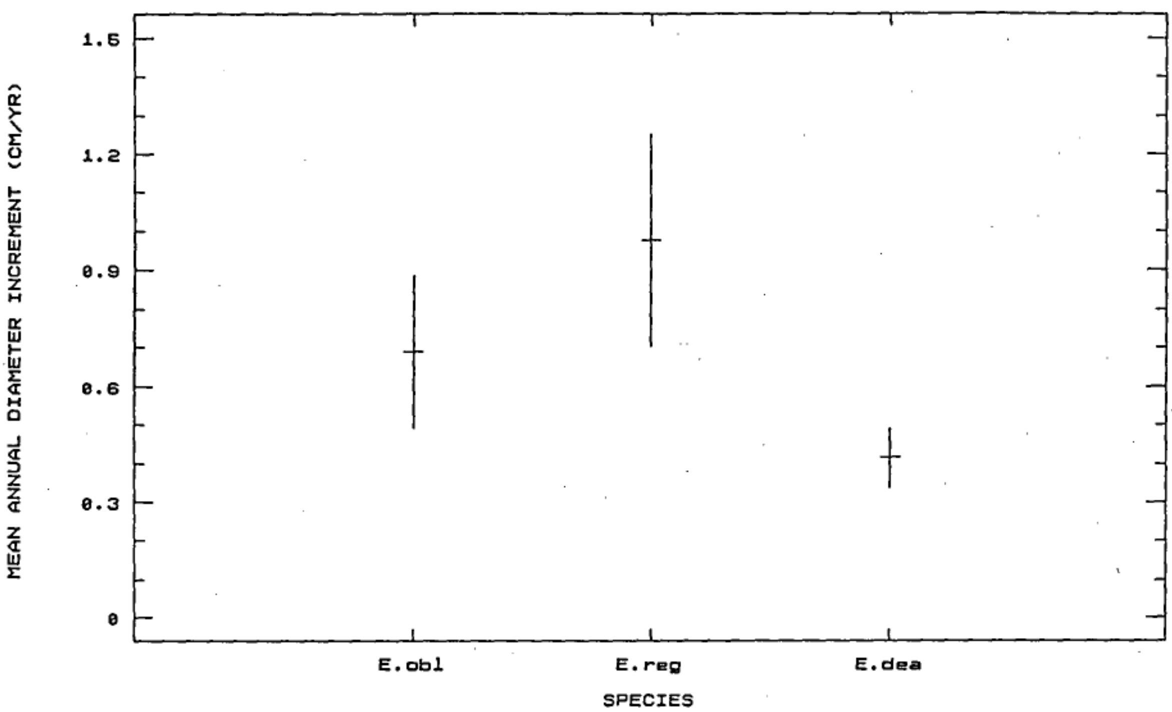
Species	Trees	Canopy class n_1,n_2,n_3,n_4	Mean dbh (cm)	Mean dbh inc. (cm y^{-1})	Current dbh inc. (cm y^{-1})
Acacia dealbata	14	0 0 7 7	20.7±6.8	0.33	0.05±0.14
Eucalyptus obliqua	16	3 3 6 4	34.5±18.4	0.69	0.16±0.26
Eucalyptus regnans	14	6 2 3 3	48.9±23.8	0.98	0.27±0.43

Table 6.8 shows that the current annual diameter increments are much lower than the mean annual diameter increments which indicates a marked slowing in diameter

increment. The plot sheets indicated that the eucalypts were showing some signs of regrowth dieback which may explain their low current increment. The low current annual diameter increment for *Acacia dealbata* is attributed to it nearing the end of its normal lifespan.

Fig. 6.2 shows the mean annual diameter increment and the 95 percent confidence limits for the three species recorded on plot 1046 at age 50 years. The figure indicates that the eucalypt species have a significantly higher mean annual diameter increment than *Acacia dealbata*.

Fig. 6.2. Mean annual diameter increment for a 50-year-old stand (plot 1046). (E.obl = *Eucalyptus obliqua*, E.reg = *E. regnans* and A.dea = *Acacia dealbata*)



Discussion

Better estimates of growth rates of special timbers are needed to determine rotation lengths which would yield appreciable quantities of special timbers. This is particularly needed for *Acacia melanoxylon* which is the most valuable special timber in mixed forest. The approach taken here may be improved by including additional plot data from other districts and by comparisons of growth of individual trees following successive measurements of plots over a substantial period. Such analyses are currently restricted because CFI data on special timbers species diameters has only been gathered since 1979. Analyses are also hampered by the lack of plot information on special timbers species less than 10 cm dbh.

Despite the sparse data it is apparent that rotations longer than 90 years will be required for mean diameters of special timbers species, at least those that are rainforest species, to reach millable sizes. The data in Table 6.7 suggest that a suitable rotation length for most special timbers species in regrowth mixed forest may need to be about two hundred years. However, this period may exceed the average life span of *Acacia melanoxylon* and will definitely exceed the life span of *Acacia dealbata*. The oldest recorded age for an *Acacia melanoxylon* tree is 210 years (Frampton 1980) although the longevity of most individuals is considered to be much less.

Acacia dealbata yields will not be increased by long rotations as this species is unlikely to survive even until the routine rotation period of 80 to 100 years. This need not be of particular concern as the species can regenerate prolifically from ground-stored seed following disturbances even after 350 to 500 years (Gilbert 1959) and currently the demand for the timber for special purposes is far exceeded by the supply (Forestry Commission 1990[a]). The commercial potential of this species is also limited by its susceptibility to severe defoliation by the fireblight beetle (Elliott and deLittle 1984).

7. OPTIONS FOR MAINTENANCE OF CONSERVATION AND SPECIAL TIMBER VALUES OF MIXED FOREST USED FOR WOOD PRODUCTION

In the past decade there has been a partial shift by forest managers, at least in the developed world, from silvicultural manipulation of natural forests primarily for maximum economic production of wood fibre towards multiple-use management where the primary aim is the maintenance of long-term ecosystem health with timber production being a by-product of that management. This shift has been brought about by a recognition that some silvicultural practices result in ecosystem simplification at the tree, stand and landscape level, and that the maintenance of biological diversity will not be achieved solely by large preserves. This change has been most evident in the Pacific North-west region of North America where a more ecological approach to forest management has been dubbed "New Forestry". The principles of New Forestry are summarised by Hopwood (1991) and include an increased understanding of:

- the structure and function of oldgrowth forests;
- the extent to which 'biological legacies' (*sensu* Franklin 1992) of surviving trees, dead stags and down logs persist through natural disturbances such as wildfire to provide diversity in the subsequent stand; and
- the habitat value of late seral stages of forest which are reduced or eliminated by intensive management.

Silvicultural practices that have been proposed or adopted at the stand level include the retention of green trees, stags and down logs within coupes and establishment of mixed species stands. Practices at the landscape level have sought to minimise forest fragmentation and to provide habitat reserves and wildlife corridors within forests used for wood production (Hopwood 1991). Harris (1984) proposed a system of long rotation islands, linked by retained riparian strips, to maintain the habitat value of late seral stages within wood production forests.

Changes to a more ecologically based approach to wood production from native forests are also taking place in Australia. Horne and Carter (1992) considered the sustainability of long-term logging in New South Wales blackbutt forests and presented a range of silvicultural options for the maintenance of biodiversity. One option promoted the restoration of oldgrowth habitat by the retention of some advanced growth trees in previously cut-over areas where oldgrowth trees were now lacking. Some other options recognised a reduction of local biodiversity for increased wood production. In Tasmania, alternative silvicultural practices are being increasingly applied to maintain ecological values. Examples of practices which may be recommended include partial cutting (instead of clearfelling), increased canopy retention, non-burning, long rotations, increased retention of habitat trees and wider streamside reserves (Taylor 1991). There has been a marked trend over the last decade towards the use of non-burning treatments in dry sclerophyll forests (McCormick and Cunningham 1989) and this has assisted the maintenance of ecological values as well as timber productivity. Although 51 percent of coupes on public land in Tasmania were harvested by partial logging techniques in 1991/92 (Forestry Commission 1992), clearfelling, burning and sowing is the most common method for re-establishment of wet eucalypt forests, including mixed forests, after logging.

Clearfelling and burning is the recommended technique for wet eucalypt forests (Gilbert and Cunningham 1972) because it achieves a dense stocking and fast growth rate of *Eucalyptus* species which form the primary timber resource. The planned rotation length of 80-100 years is sufficiently similar to the natural occurrence of wildfire in wet sclerophyll forest of 20-100 years (Jackson 1968) to assume that there will not be a major shift in species composition by silvicultural treatment. This is supported, at least in the short term, by the floristic study of clearfelled wet eucalypt forest by Dickinson and Kirkpatrick (1987). However, the same rotation length applied to mixed forest represents a marked shift from the natural wildfire frequency of 100 to 400 years and has a more significant effect on species composition of the regenerated forest compared to mature examples of that forest type. Chapter 3 indicates that after a single treatment of clearfelling, burning and eucalypt sowing, the vascular plant floristics of silvicultural regeneration were sufficiently similar to wildfire regeneration of mixed forest to assume that, in the absence of further logging or fires, the silvicultural regeneration could become mixed forest and eventually rainforest. Therefore it is possible that current regeneration methods are consistent with the re-establishment of mixed forest given sufficient time for the wet sclerophyll understorey species to senesce and for rainforest species to become prominent in the understorey. This period is uncertain but from the work of Gilbert (1959) would be a minimum of 100 years. A longer period, say 200 years, would allow rainforest species to: attain larger sizes, produce greater amounts of seed; increase in density through autogenic replacement (Read and Hill 1988); and increase their ability to survive either as individuals or as seed following a clearfell, burn and sow treatment. Therefore the critical factor in the silvicultural perpetuation of mixed forest may be rotation length rather than regeneration treatment. However, it is clear that there are other regeneration methods which will allow much greater initial regeneration of rainforest species than by clearfelling and burning. It should be recognised that even with the application of the most effective rainforest regeneration methods and long rotations there may be an increased potential for disease and wildfire to disrupt successional processes in wood production forests due to the presence of roads and the use of prescribed burning.

Chapter 2 of this study indicates that there are about 39,000 ha of mixed forest with a mature myrtle understorey and 54,000 ha of eucalypt forest with an understorey of 'secondary species' on State forest in Tasmania. A subset of these areas are classed as Multiple-Use Forests (Forests and Forest Industry Council 1990) and are being used for several purposes including wood production. It is desirable to maintain mixed forest outside reserves for their conservation, aesthetic and special timber values. This chapter considers some options for the perpetuation of mixed forest within areas used for wood production.

Silvicultural options for mixed forest

Various silvicultural treatments for mixed forest and rainforest have been proposed (e.g. Combined Environment Groups 1988, Forestry Commission 1990[a], Hickey and Felton 1991, Barker 1992, Allen 1992) and are summarised in Table 8.1.

Only treatment 1 is practised routinely although treatment 2 is essentially the same and merely requires a longer rotation. Treatment 3 has been applied in a no-burn trial at Smithton (McCormick 1982) and in a number of lightly burnt coupes sampled by Jordan *et al.* (1992). Eucalypt productivity in wetter areas of Tasmania is generally better on burnt sites compared with unburnt sites (Lockett and Candy 1984) although Jordan *et al.* (1992) observed better early eucalypt growth on unburnt, infertile sites than on similar burnt sites. KIng *et al.* 1993 tested independently the effects of burning, slash accumulation and soil disturbance on regeneration of *Eucalyptus regnans* in Victoria. Although burnt seedbeds provided optimum conditions for eucalypt seedling establishment, topsoil disturbance due to logging also provided adequate conditions for seedling establishment. Heavy slash accumulation was found to inhibit adequate regeneration.

Treatment 4 is essentially the same as treatment 3 and merely requires a longer rotation. Similar treatments to 5 and 6 have been applied in a number of trials in rainforest (Hickey and Felton 1991). Treatments 7 and 8 have not been tested although treatment 8 has been informally applied in some areas, e.g. south of Mawbanna in north-western Tasmania, prior to the introduction of clearfelling and a market for rainforest species pulpwood. The forest is visually unattractive and appears to have low productivity. Treatment 9 has recently been tested in a trial in Geeveston District (Allen 1992) and appears unsuitable for oldgrowth forest with a dense rainforest understorey.

Table 7.1 Silvicultural treatments for regeneration of tall mixed forest (modified from Forestry Commission (1990[a])).

Treatment	Rotation (years)	Advantages	Problems
1. Clearfell, burn, and sow to eucalypts (standard treatment).	80-100	Cheap, proven system; high eucalypt productivity	Sparse rainforest species regeneration; low special timbers yield; potential loss of rainforest tree species after second cycle?
2. Clearfell, burn, and sow to eucalypts (long rotation).	200-300	Cheap system; adequate rainforest regeneration; high quality eucalypt and special timbers supply.	Reduced eucalypt yield.
3. Clearfell, burn, and sow to eucalypts (no burn/light burn).	80-100	Cheap system; moderate rainforest regeneration.	Low eucalypt density and yield; high fire risk; low special timbers yield; potential loss of rainforest tree species after repeated cycles?
4. Clearfell, burn, and sow to eucalypts (no burn/light burn)	200-300	Cheap system; good rainforest regeneration; high quality special timbers supply.	Low eucalypt density and yield; high fire risk.
5. Seed tree retention. Log and leave 20 rainforest trees ha^{-1} and 10 eucalypt trees ha^{-1} (no burn).	200-300	Cheap system; excellent rainforest regeneration; high quality special timbers supply.	Low eucalypt density and yield; high fire risk.
6. As for 5 plus thinning (no burn).	100-150	As for 5 but with reduced rotation.	Very expensive; high fire and myrtle wilt risk.
7. Selective logging of eucalypt and rainforest trees (down to a 40 cm diameter limit; no burn).	50-100 cycle	Small continuing supply of eucalypt and special timbers.	Low productivity; high fire and myrtle wilt risk; difficult supervision.
8. Log all eucalypts, selectively log rainforest (no burn).	50-100 cycle	Low initial impact of logging	Low yield; high levels of myrtle wilt; high fire risk; difficult supervision.
9. Group selection (clearfell small patches of about 1 ha; burning optional).	100-300	Lower impact of logging.	Expensive and dangerous; high myrtle wilt in retained groups; low eucalypt yield; high browsing problem.

The long rotation treatments would all result in a reduction in the yield of eucalypt timber compared with treatment 1, as eucalypt growth rates decline in the latter part of the rotation. Treatments 3-8 would result in lower eucalypt densities than achieved by treatments 1 and 2. This would be only slightly offset by a modest yield of special timber species. An even greater reduction would result in the next few decades if the harvesting of extensive areas of oldgrowth mixed forest currently scheduled for logging were delayed to achieve a small continuing cut of special timbers species from these forests (G. Bradbury pers. comm.).

The selective logging treatments, 7-9, would be inappropriate for rainforest areas with a high proportion of *Nothofagus cunninghamii* due to excessive levels of myrtle wilt (Packham 1991). Selective logging in podocarp forest in New Zealand resulted in net stand decrement in the first few years after logging (Smale *et al.* 1985) due to windthrow and logging damage and a similar, or worse, result can be expected from selective logging in *Nothofagus cunninghamii* forest.

It would be unwise to practise burn and no-burn treatments in adjacent areas unless they were well separated temporally. If it was considered desirable to apply long rotations to a proportion of wet eucalypt forest in a wood production zone where the standard regeneration technique was treatment 1, then treatment 2 would probably be the most appropriate silvicultural regime. If, however, it was intended to manage mixed forest for special timbers in areas adjacent to rainforest, as advocated by the Forests and Forest Industry Council (1990), then treatments 4 and 5 would be more appropriate. The relative merits of these treatments could only be assessed after a number of years of operational trials.

The consideration, and application of alternative treatments in Tasmanian mixed forests will attract similar criticism to that surrounding the application of New Forestry practices in the Pacific North-west. These have been summarised by Atkinson (1990) as:

- additional harvesting costs;
- increased danger to timber fellers due to the presence of standing dead wood;
- reduced regeneration and growth of commercial timber species;
- increased risk of soil compaction following repeated incursions into partially-cut stands; and
- coupes look like a mess and are no aesthetic improvement on clearfelled areas.

There is some validity in all these criticisms and, in Tasmania at least, they may all be subordinate to the criticism of increased fire hazard. There is little doubt that New

Forestry practices represent a sub-optimal treatment in terms of timber production. However their application may allow the retention of a far greater array of non-wood values while still allowing the financial benefits of some wood production and increased access to the forest for other uses.

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Appendix 1. Sample legends of photo-interpreted maps from 1949, 1969 and 1991.

1. Legend from PICTON 87/15 (1: 15,840) compiled in 1949 from 1947 and 1948 photography

Photo-interpreted types

E	Eucalypt forest
E1	Mature eucalypt forest average height of dominants above 180 feet
E2	Mature eucalypt forest average height of dominants from 136-180 feet
E3	Mature eucalypt forest average height of dominants from 90-135 feet
E4	Mature eucalypt forest average height of dominants less than 90 feet
ER	Eucalypt regrowth
ERg	Eucalypt seedling and sapling regrowth
ERp	Eucalypt pole regrowth
M	Myrtle
MM	Myrtle and associated species
V	Agricultural and grazing land
W	Wasteland
Wg	Button grass, heathy plain
Wm	Mountain moor
Wr	Bare ground or rock
T	Understorey species
S	Scrub
K	Bracken
f/d	Fire-damaged
O/m	Overmature
C/o	Cut over

Density classes

A	Dense forest
B	Semi-dense forest
C	Open forest
D	Very open forest
F	Scattered trees

2. Legend from PICTON 94/11 (1: 15,840) compiled in 1969 from 1966 photography

Photo-interpreted types

Mature Eucalypt

Height Classes		*Density Classes*	
E1*	average height above 250 ft	a	70-100 percent crown cover
E1	average height from 180-250 ft	b	40-70 percent crown cover
E2	average height from 135-180 ft	c	20-40 percent crown cover
E3	average height from 90-135 ft	d	5-20 percent crown cover
E4	average height from 50-90 ft	f	less than 5 percent crown cover
E5	average height less than 90 ft		

Regrowth Eucalypt

Height Classes		*Density Classes*	
ER1	up to 50 ft	a	90-100 percent crown cover
ER2	from 50-90 ft	b	70-90 percent crown cover
ER3	from 90-120 ft	c	50-70 percent crown cover
ER4	from 120-145 ft	d	10-50 percent crown cover
ER5	from 145-165 ft	f	1-10 percent crown cover
ER6	above 165 ft		

Rainforest

M1	mature myrtle with average height above 120 ft
M2	mature myrtle with average height from 80-120 ft
M3	mature myrtle less than 80 ft
Mr	myrtle regrowth

Other vegetation

T	secondary species
T(W)	wattle
V	cultivation and pasture
Vz	rough grazing
S	scrub
K	bracken
W	wasteland
Wg	button grass or heathy plain
Wm	mountain moor
Wr	bare ground or rock
Pr	pine plantation
f/d	fire damaged
o/m	over mature
c/o	cut over
(p)	patches

3. Legend from SUMAC 3244 (1: 25,000) compiled in 1991 from 1984 photography

<u>Photo-interpreted types</u>

Mature Eucalypt

Height Classes		*Density Classes*	
E1*	average height more than 76 m	a	70-100 percent crown cover
E1	average height from 55-76 m	b	40-70 percent crown cover
E2	average height from 41-55 m	c	20-40 percent crown cover
E+3	average height from 34-41 m	d	5-20 percent crown cover
E-3	average height from 27-34 m	f	less than 5 percent crown cover
E4	average height from 15-27 m		
E5	average height less 15 m		

Regrowth Eucalypt

Height Classes		*Density Classes*	
ER1	average height less than 15 m	a	90-100 percent crown cover
ER2	average height from 15-27 m	b	70-90 percent crown cover
ER3	average height from 27-37 m	c	50-70 percent crown cover
ER4	average height from 37-44 m	d	10-50 percent crown cover
ER5	average height from 44-50 m	f	1-10 percent crown cover
ER6	average height more than 50 m		

Regeneration

N	naturally seeded
A	artificially seeded
N&A	part natural and part sown
P	planted
T	thinned
W	wildfire
(72)	year of regeneration (1972)
2	original oldgrowth height class (E2)
X	original oldgrowth height class unknown

Rainforest

M+	large crowned rainforest
M-	small crowned shorter rainforest
MR1	myrtle regrowth less than 15 m
Mr	myrtle regrowth 15 m or greater

Other vegetation

T	secondary species	Wm	mountain moor
T(W)	wattle	Wr	bare ground or rock
V	cultivation and pasture	Pr	pine plantation
Vz	rough grazing	f/d	fire damaged
S	scrub	o/m	over mature
K	bracken	c/o	cut over
W	wasteland	(p)	patches
Wg	button grass or heathy plain		

Appendix 2. A review of unpublished studies on rainforest tree species regeneration following logging of mixed forest in Tasmania.

Some unpublished studies on rainforest tree species regeneration following logging of mixed forest exist have been undertaken by the Forestry Commission, Tasmania and Australian Newsprint Mills (ANM) Forest Management Division. These include:

- a report of a survey of rainforest tree species regeneration in eucalypt regeneration coupes in Smithton District (Blakesley 1978);

- field books and maps from a survey of rainforest tree species regeneration in eucalypt regeneration coupes in Geeveston and Norfolk (within the ANM Concession) Districts (S. Calais unpublished data);

- brief reports of a trial in Smithton District which attempted to regenerate mixed forest without a regeneration burn (McCormick 1982, Neyland 1983); and

- an assessment of information from permanent (ANM) plots in 25-30-year-old silvicultural regeneration in the Florentine Valley (Heathcote 1985).

Information from each of these sources is summarised in this appendix.

Regeneration surveys in Smithton District

Blakesley (1978) surveyed 12 regeneration coupes, between 2 and 16 years old, and three areas burnt by wildfires, 12 to 24 years previously, which were all either mixed forest or rainforest prior to the disturbance. Results of the surveys, reported by Felton and Lockett (1983), showed that 19-62 percent of $4m^2$ plots in the logged coupes and 60-87 percent of plots in the wildfire areas were stocked with either *Acacia melanoxylon, Nothofagus cunninghamii, Eucryphia lucida, Atherosperma moschatum* or *Phyllocladus aspleniifolius.*

Blakesley made the following general observations:

- very few rainforest seedlings are present on large burnt areas which lack remnant seed sources, except for *Acacia melanoxylon* [a doubtful rainforest species], occasional *Phyllocladus aspleniifolius* seedlings and small *Atherosperma moschatum* cotyledonary seedlings;

- substantial *Nothofagus cunninghamii* and *Eucryphia lucida* regeneration occurs on snig tracks where the seedlings have had some protection from the regeneration burn;

- *Atherosperma moschatum* cotyledonary seedlings occur widely in regeneration areas but few seedlings become established;

- logged unburnt areas generally have a good stocking of rainforest tree species regeneration;

- heavy browsing of seedlings of *Acacia melanoxylon*, and probably of *Atherosperma moschatum,* occurs in regeneration areas; and

- Cull-felling of the rainforest understorey for fuel preparation reduces the occurrence of rainforest species regeneration because it depletes seed and shade sources and results in hotter regeneration burns.

Blakesley concluded that the ideal treatment for rainforest tree species regeneration following logging of mixed forest was to retain rainforest cull (unmerchantable) trees and leave the site unburnt. He noted this would probably lead to inadequate eucalypt regeneration and suggested a suitable compromise treatment may be to leave groups of rainforest cull trees as seed and shade trees and carry out a less intense regeneration burn.

Regeneration surveys in Geeveston District and the ANM Concession

Satwant Calais carried out regeneration surveys, using $4m^2$ plots, of rainforest tree species in 14 regeneration coupes, aged from 2 to 16 years, in Geeveston District and 13 regeneration coupes, between 2 and 17 years old, in the ANM Concession (within Norfolk District) in 1976-77. The raw data were analysed and reported by Hickey and Savva (1992). The analysis showed that rainforest tree species regeneration was common after clearfelling and burning of mixed forest in both study areas. Plots which contained at least one rainforest species were classed as stocked. The stocking of rainforest species was reduced by increased fire intensity. The density of regeneration on stocked unburnt plots was not significantly greater than on stocked plots burnt by cool fires, but unburnt stocked plots contained significantly more seedlings than stocked plots burnt by moderate or hot fires. Better rainforest species regeneration was found where the regeneration burn had either failed or was of low intensity. Only 50 and 55 percent respectively of plots in the two study areas had been subjected to a moderate or hot fire which suggests that, in many cases, silvicultural regeneration burns produce a mosaic of microsites.

Rainforest tree species regeneration was highly clumped with the best regeneration occurring within 20 m of a seed source. However, regeneration was not confined to the vicinity of seed sources. Some *Nothofagus cunninghamii*, *Eucryphia lucida* and *Atherosperma moschatum* regeneration resulted from basal sprouts as well as from seed. However, no vegetative regeneration was observed for *Phyllocladus aspleniifolius* or *Acacia melanoxylon.*

Mixed forest regeneration trial in Smithton District

In May 1981 a logged and unburnt 14 ha portion of Holder Compartment 9 (Arthur River 1:100,000 map reference 578480) was sown with eucalypt seed at a rate of 0.5 kg ha^{-1} in order to monitor the regeneration of eucalypts and rainforest tree species in the absence of a regeneration burn (McCormick 1982). The site had been logged for sawlogs (all species) in 1979 and subsequently for eucalypt pulpwood in 1980. Results of a stripline assessment of the remaining live cull trees are shown in Table 1.

Table 1. Density of cull (unmerchantable) trees at Holder Compartment 9.

Species	Trees ha^{-1}
Atherosperma moschatum	10
Eucalyptus spp.	3
Eucryphia lucida	133
Nothofagus cunninghamii	68
Phyllocladus aspleniifolius	20
Total	234

Regeneration surveys of the trial were carried out in March 1982 and December 1983 (Neyland 1983) and the results are shown in Table 2. The survey method was to record the stocking (presence/absence) on $4 m^2$ plots located at 20 m intervals on subjectively located striplines. The same striplines were not remeasured so the results are not directly comparable although a clear trend is evident.

Table 2. Regeneration survey results for the mixed forest regeneration trial at Holder Compartment 9.

Species	% $4 m^2$ plots stocked	
	March 1982	December 1983
Acacia melanoxylon	4	3
Atherosperma moschatum	0	0
Eucalyptus spp.	20	16
Eucryphia lucida	33	62
Nothofagus cunninghamii	22	57
Phyllocladus aspleniifolius	8	16
Stocked (any species)	61	78

The surveys indicate a substantial initial stocking of *Eucryphia lucida* and *Nothofagus cunninghamii* which further increased following subsequent seed falls. Seedfall of *E. lucida* is fairly consistent from year to year but *N. cunninghamii* has heavy seedfalls every two or three years with low seedfalls in intervening years (Hickey *et al.* 1982). A heavy seedfall of *N. cunninghamii* occurred in 1982. The stocking of eucalypts is low and would be considered unsatisfactory by current standards (see Forestry Commission 1991). However, if the management aim is to regenerate a mixed forest of both eucalypts and rainforest trees then the total regeneration of tree species may be adequate. Initial regeneration of *Acacia melanoxylon* and *Atherosperma moschatum* was very poor, possibly due to marsupial browsing, although regeneration could be expected to improve as a protective cover of vegetation develops. Unfortunately subsequent monitoring was not carried out and the trial was burnt by an extensive wildfire in 1988. One of the major criticisms of 'no-burn' treatments for regeneration of wet forests is that large amounts of highly flammable logging slash pose an excessive fire-hazard at least for the first decade following logging. However, it should be noted that many areas of silvicultural regeneration less than 10 years old and established after regeneration burns were also consumed in the same wildfire and in an adjacent fire in 1984 (Jennings 1991).

Regrowth assessment in the Florentine Valley

Heathcote (1985) carried out an assessment of data from 95 0.02 ha plots located in silvicultural regeneration established between 1955 and 1960 (prior to the routine adoption of the clearfell, burn and sow technique) in the Florentine Valley. No information on the pre-logging forest type was collected. The silvicultural techniques used for regeneration are uncertain but it is likely that most areas were regenerated using low intensity slash burns with seed shed from retained eucalypt seed trees. Table 3 summarises information on species presence from the survey.

Table 3.3 Percentage of plots in 25-30-year-old silvicultural regeneration in the Florentine Valley according to species presence (after Heathcote 1985).

Trees	Percent
Acacia dealbata	78.8
Acacia melanoxylon	28.3
Atherosperma moschatum	22.1
Eucalyptus delegatensis	60.3
Eucalyptus regnans	81.2
Eucalyptus viminalis/dalrympleana	6.1
Eucryphia lucida	11.1
Nothofagus cunninghamii	43.0
Olearia argophylla	32.0
Phyllocladus aspleniifolius	7.4
Pomaderris apetala	46.7
Ferns	
Dicksonia antarctica	50.4
Pteridium esculentum	65.2

Heathcote reported that regeneration of rainforest species in silvicultural regeneration was widespread and that growth rates of *Acacia dealbata* were sometimes as high as those of the fastest growing eucalypts.

Appendix 3. Archived and current photo-interpreted maps of wildfire sites.

Site Number: 13, 14 & 26

Grid Reference: E341090 N^{5}432980
E341000 N^{5}432950
E341050 N^{5}433120

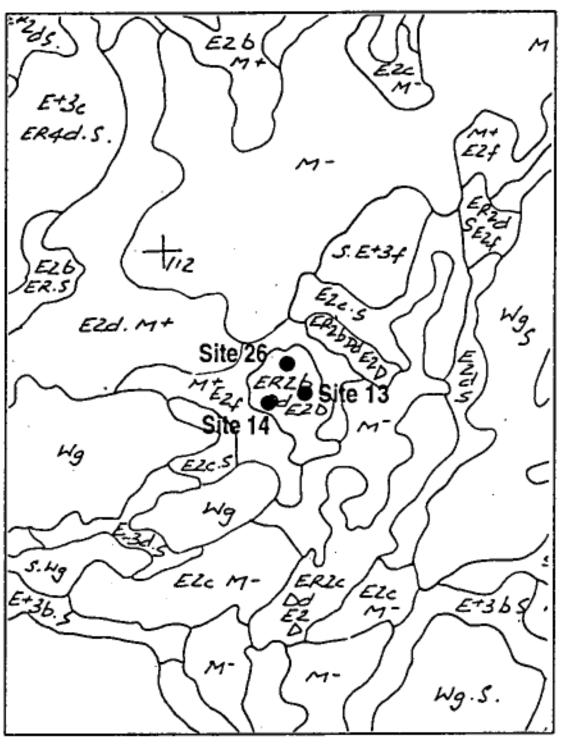
District: Smithton

Locality: South of Lost Hill

Wildfire Date: 1961

Reference: Davis (1981)

Comments:

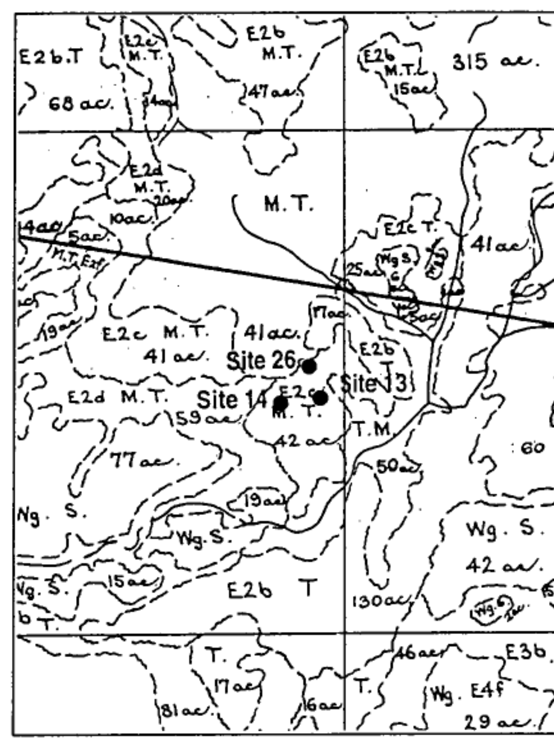


Map Name: Beryl

Map Number: 3443

Scale: 1 : 25 000

Date of Photography: 1989



Map Name: Trowutta

Map Number: 27/9&13

Scale: 1 : 25 000 (from 1:23 760)

Date of Photography: 1952-53

Site Number: 15, 30, 31 & 32

Grid Reference: E483300 N⁵219050
E483530 N⁵217830
E483550 N⁵218530
E483300 N⁵218950

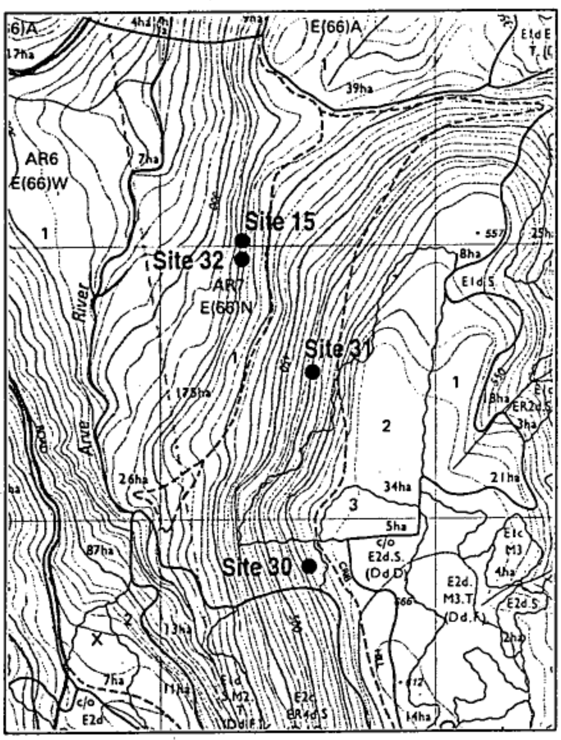
District: Geeveston

Locality: Crib Hill Road

Wildfire Date: March 1966

Reference: Forestry Commission (1967)

Comments: escaped regeneration burn

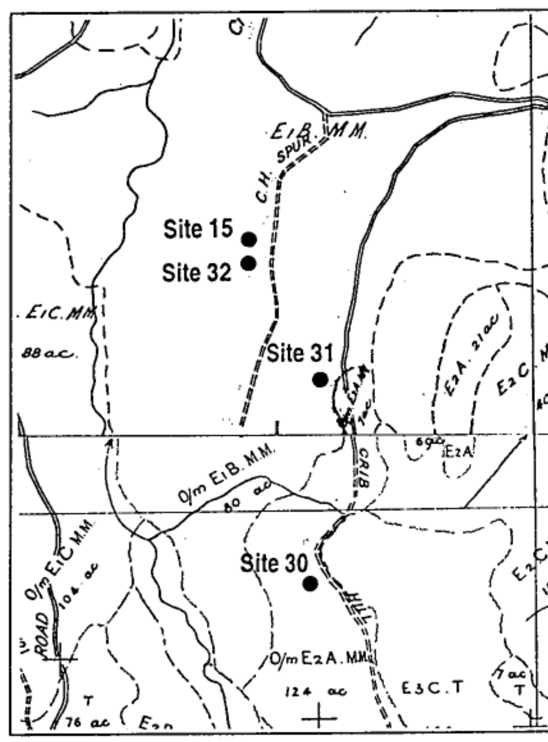


Map Name: Waterloo

Map Number: 4821

Scale: 1 : 25 000

Date of Photography: 1984



Map Name: Picton

Map Number: 87/11 & 87/15

Scale: 1 : 25 000 (from 1:15 840)

Date of Photography: 1947

Site Number: 29

Grid Reference: E512400 N^{5}254130

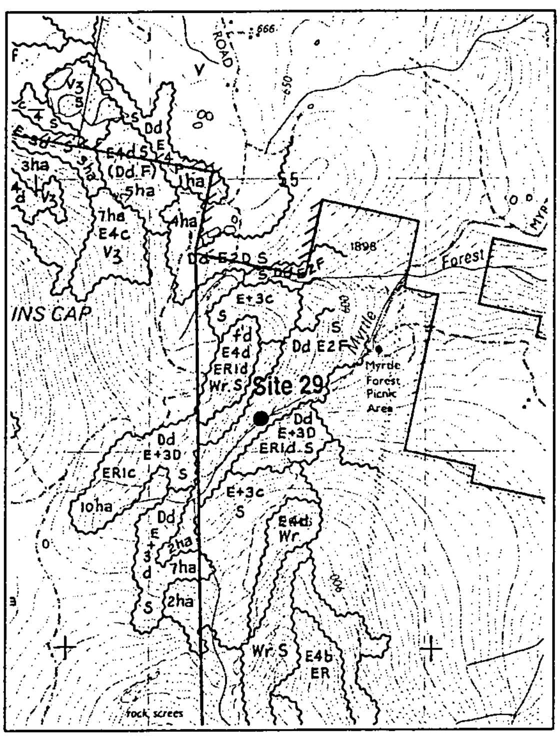
District: Norfolk

Locality: Collinsvale

Wildfire Date: February 1967

Reference: Forestry Commission (1967)

Comments: (site not used for analysis due to an absence of comparable old growth forest or silvicultural regeneration)



Map Name: Collinsvale

Map Number: 5025

Scale: 1 : 25 000

Date of Photography: 1982

No Previous PI Map Exists

Map Name:

Map Number:

Scale:

Date of Photography:

Site Number: 33, 34 & 35

Grid Reference: E332980 N⁵446460
E332980 N⁵446610
E332980 N⁵446730

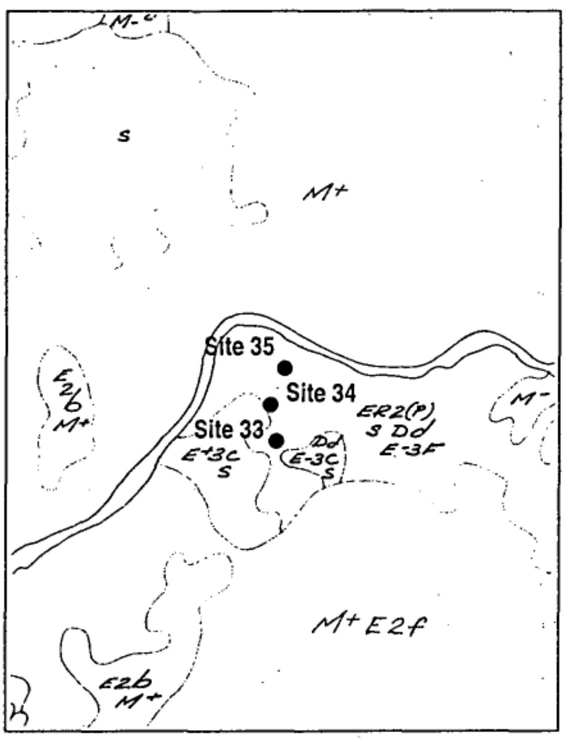
District: Smithton

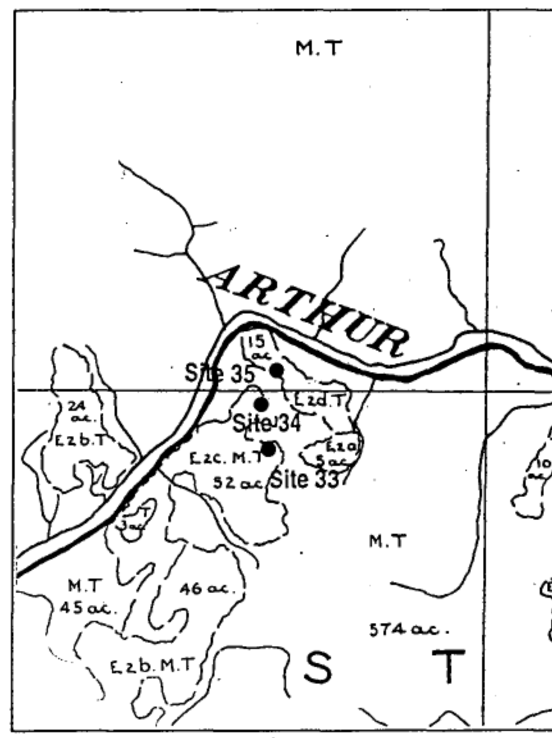
Locality: Sumac Road

Wildfire Date: February - March 1966

Reference: Forestry Commission (1966)

Comments: part of the 1966 Roger River West fire





Map Name: Sumac

Map Number: 3244

Scale: 1 : 25 000

Date of Photography: 1988-89

Map Name: Trowutta

Map Number: 27/1&5

Scale: 1 : 25 000 (from 1:23 760)

Date of Photography: 1947

Site Number: 36

Grid Reference: E347700 N[5]453240

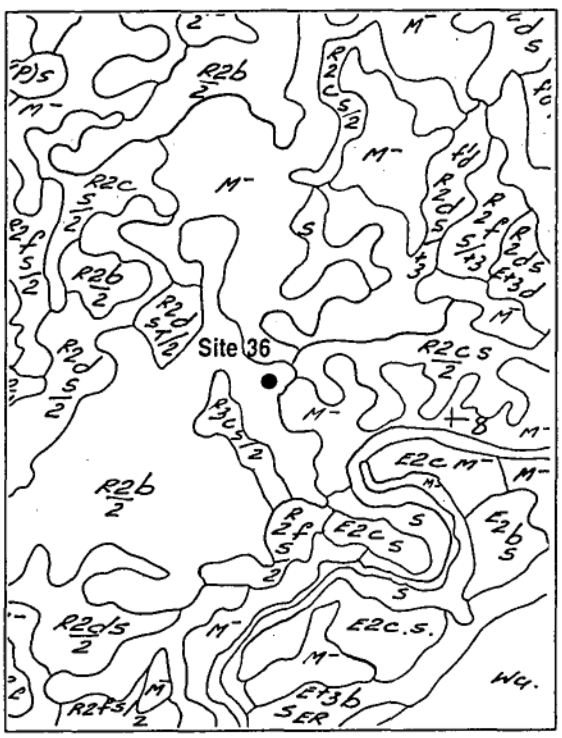
District: Smithton

Locality: Trowutta

Wildfire Date: January - February 1962

Reference: Forestry Commission (1962)

Comments: light selective logging (pre-fire) near site

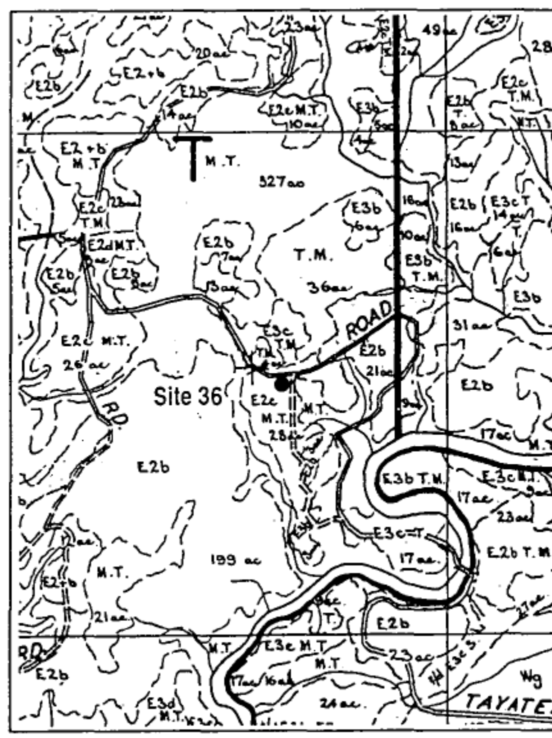


Map Name: Tayatea

Map Number: 3445

Scale: 1 : 25 000

Date of Photography: 1989



Map Name: Trowutta

Map Number: 27/2&6

Scale: 1 : 25 000 (from 1:23 760)

Date of Photography: 1952-53

Site Number: 38

Grid Reference: E525480 N^{5}190650

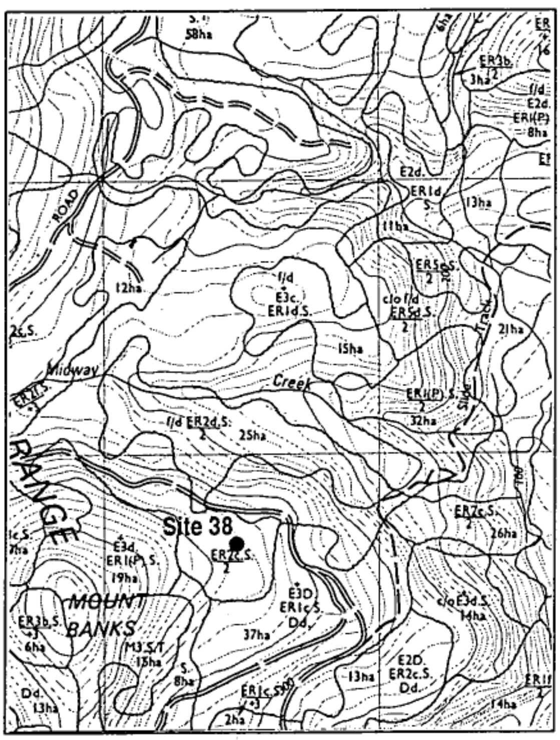
District: Geeveston

Locality: Lockleys Road (Bruny Is.)

Wildfire Date: February 1967

Reference: Forestry Commission (1967)

Comments: light selective logging (pre-fire) near site

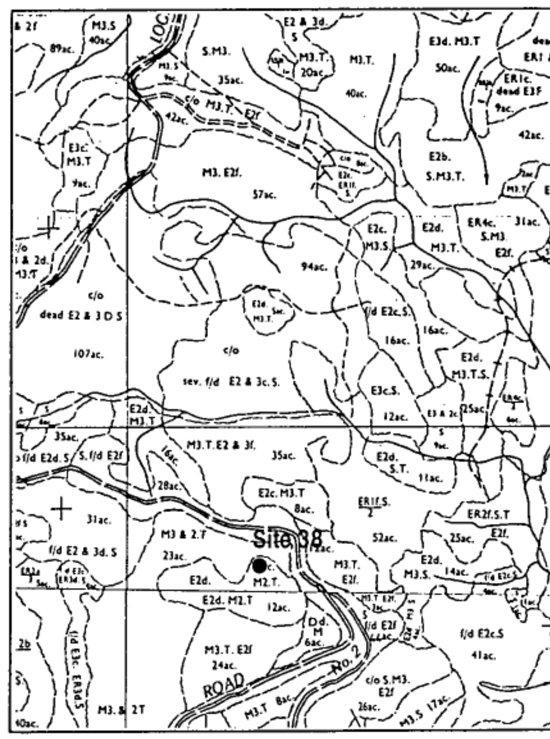


Map Name: Fluted Cape

Map Number: 5219

Scale: 1 : 25 000

Date of Photography: 1984



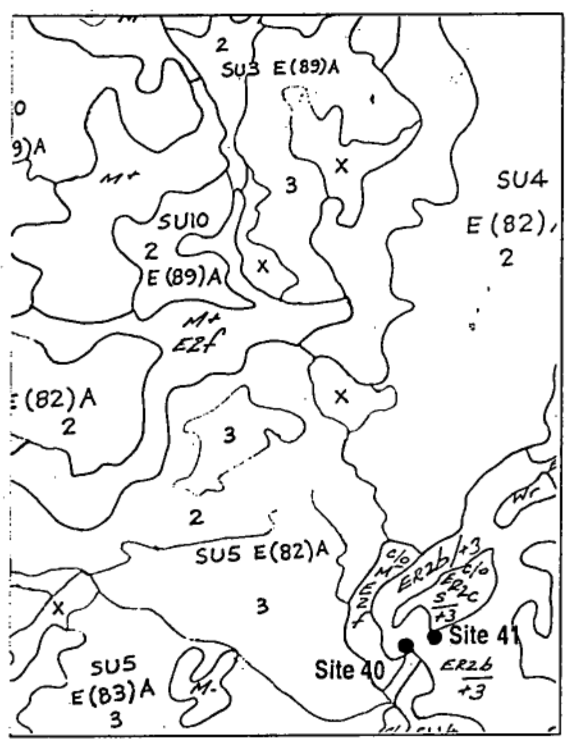
Map Name: Picton

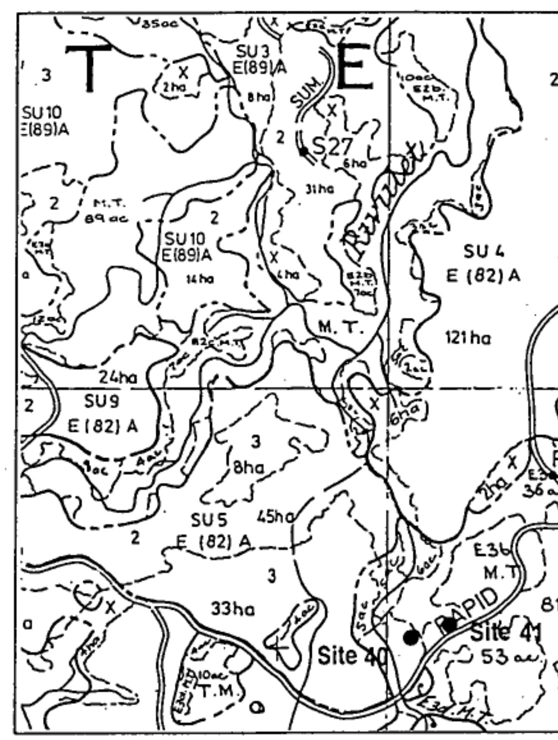
Map Number: 94/11

Scale: 1 : 25 000 (from 1:15 841)

Date of Photography: 1966

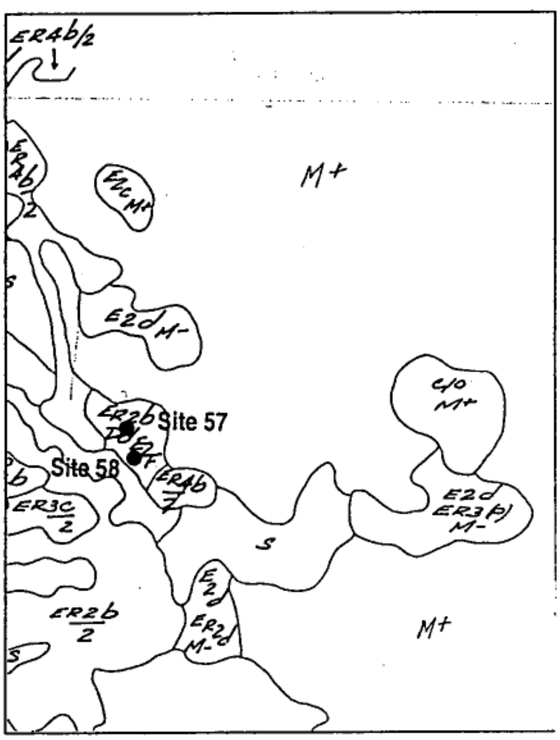
Site Number:	40 & 41	**Wildfire Date:**	1961
Grid Reference:	E339460 $N^{5}440250$ E339500 $N^{5}440350$	**Reference:**	Davis (1981)
		Comments:	
District:	Smithton		
Locality:	Rapid River Road		

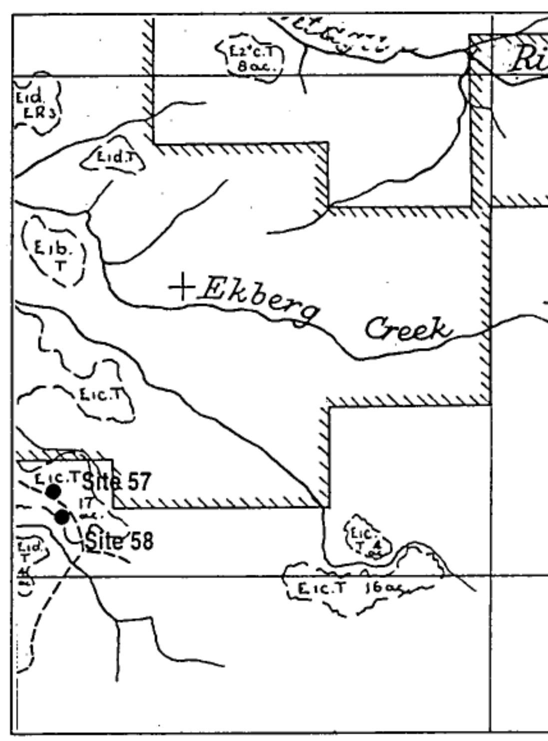




Map Name:	Sumac	**Map Name:**	Trowutta
Map Number:	3244	**Map Number:**	27/9&13
Scale:	1 : 25 000	**Scale:**	1 : 25 000 (from 1:23 760)
Date of Photography:	1988-89	**Date of Photography:**	1947

Site Number:	57 & 58	**Wildfire Date:**	February - March 1966
Grid Reference:	E332100 N⁵448800 E332150 N⁵448750	**Reference:**	Forestry Commission (1966)
District:	Smithton	**Comments:**	light selective logging (pre-fire) near sites
Locality:	Roger River West		





Map Name:	Sumac	**Map Name:**	Trowutta
Map Number:	3244	**Map Number:**	27/1&5
Scale:	1 : 25 000	**Scale:**	1 : 25 000 (from 1:23 760)
Date of Photography:	1988-89	**Date of Photography:**	1947

Site Number: 66

Grid Reference: E484660 N[5]221030

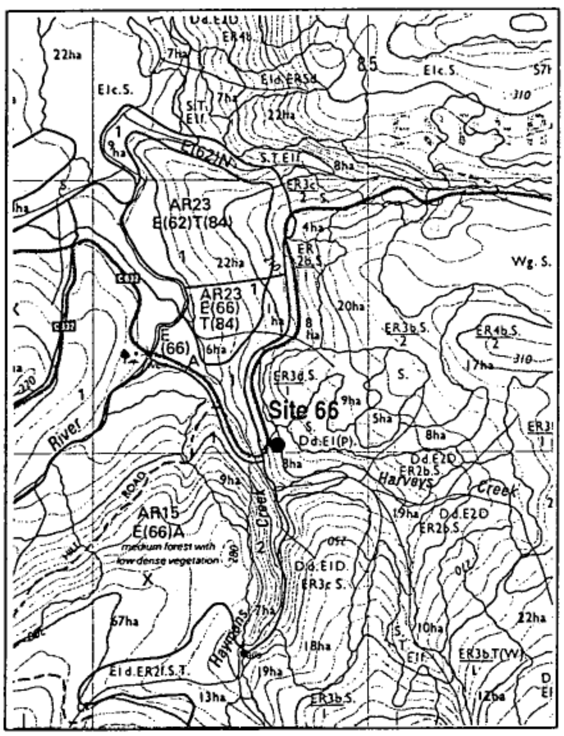
District: Geeveston

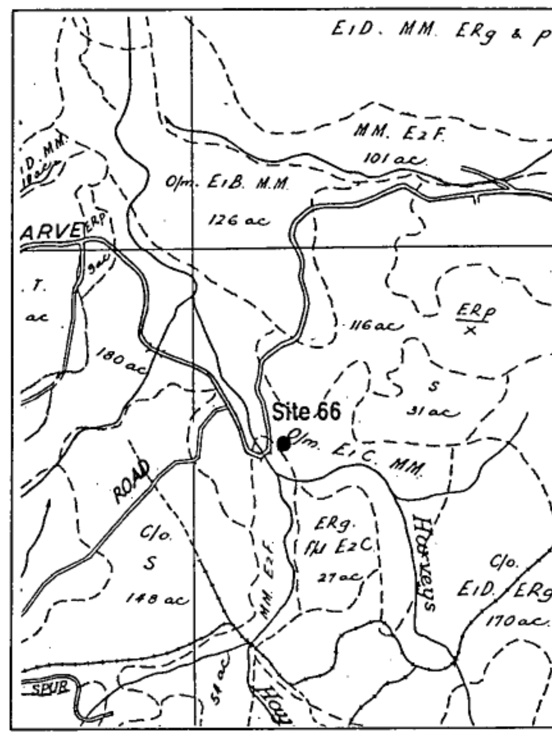
Locality: Crib HIll Road

Wildfire Date: March 1966

Reference: Forestry Commission (1967)

Comments: escaped regeneration burn





Map Name: Geeveston

Map Number: 4822

Scale: 1 : 25 000

Date of Photography: 1984

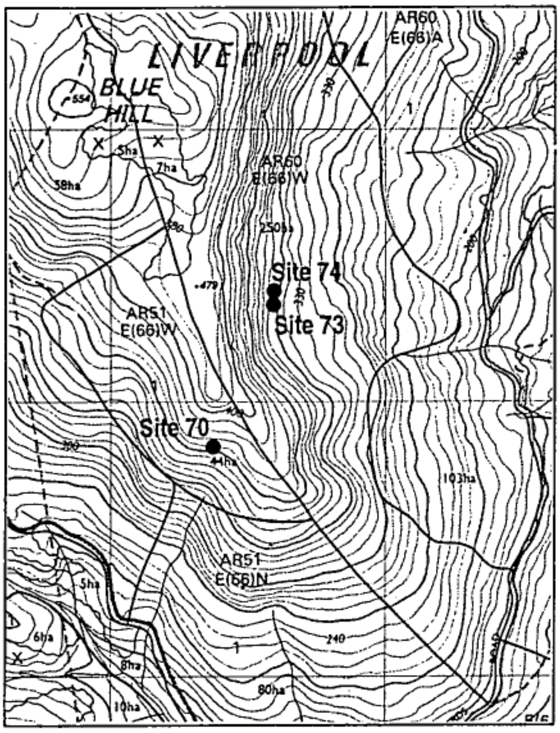
Map Name: Picton

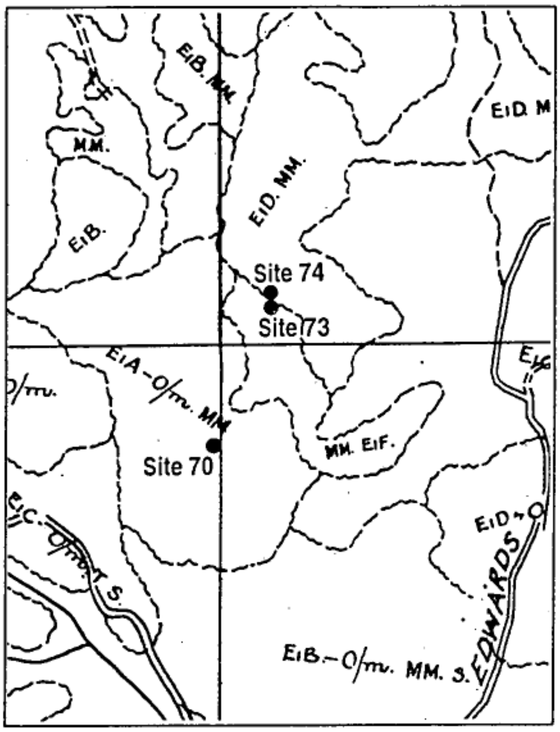
Map Number: 87/11

Scale: 1 : 25 000 (from 1:15 840)

Date of Photography: 1947

Site Number:	70, 73 & 74	**Wildfire Date:**	November - December 1966
Grid Reference:	E482380 N[5]226830 E482600 N[5]227350 E482600 N[5]227390	**Reference:**	Forestry Commission (1967)
District:	Geeveston	**Comments:**	
Locality:	Blue Hill (Arve Road)		





Map Name:	Geeveston	**Map Name:**	Picton
Map Number:	4822	**Map Number:**	87/6&10 87/7&11
Scale:	1 : 25 000	**Scale:**	1 : 25 000 (from 1:31 680)
Date of Photography:	1984	**Date of Photography:**	1947

Site Number: 75, 76, 93, 94, 100 & 101

Grid Reference:
E458440 N[5]277890
E458510 N[5]277730
E458500 N[5]277730
E458600 N[5]277600
E458750 N[5]277050
E458750 N[5]277150

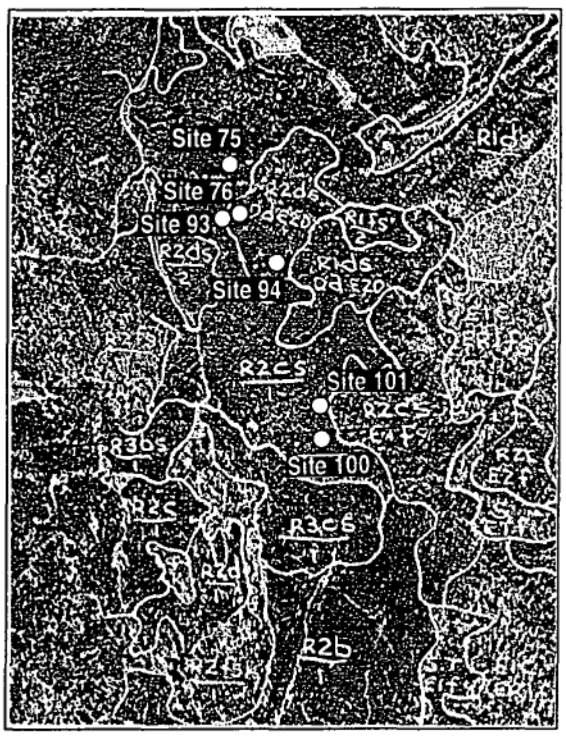
District: Norfolk

Locality: Westfield Quarry
(Florentine Valley)

Wildfire Date: 1966

Reference: M. White (pers. comm.)

Comments:

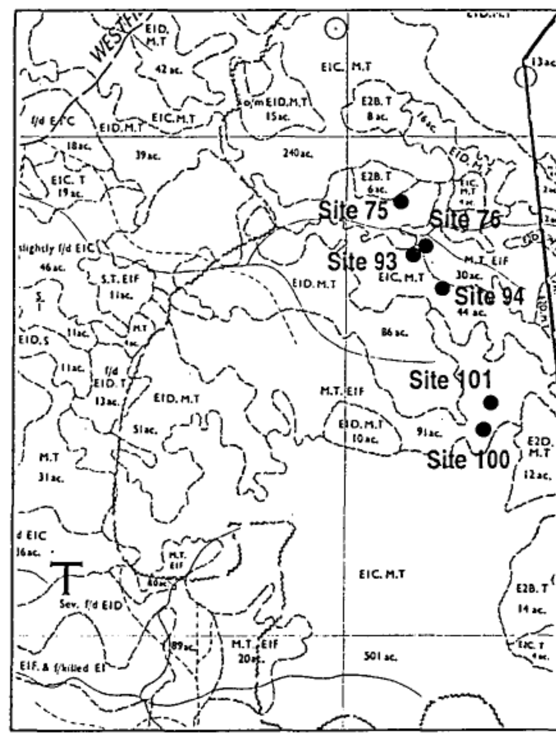


Map Name: Aerial Photo
Florentine-Maydena

Map Number: 1168-85 M931
Run 67

Scale: 1 : 20 000

Date of Photography: 1991



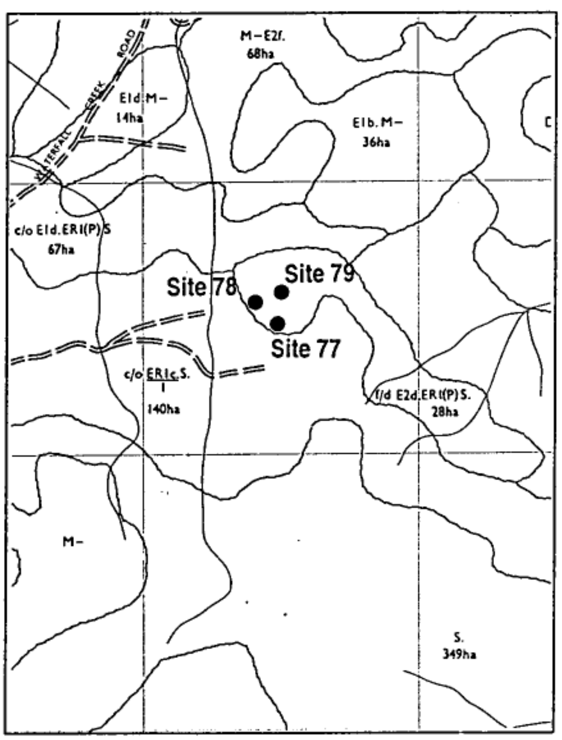
Map Name: ANM
Florentine Concession

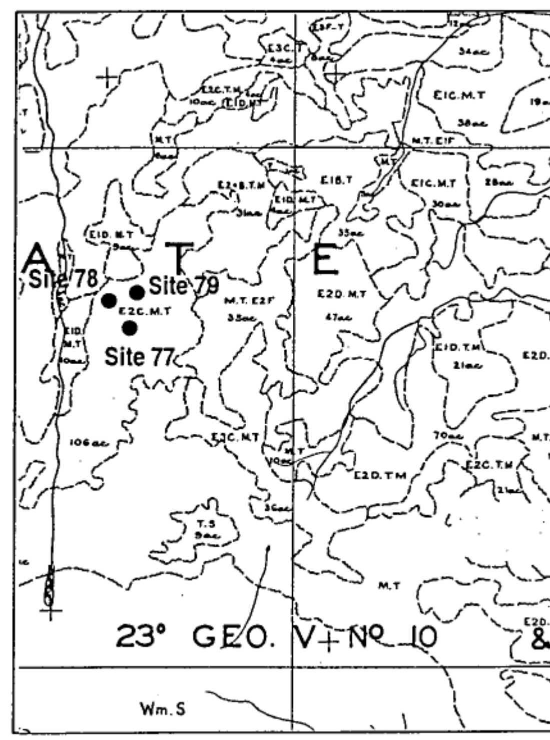
Map Number: H12

Scale: 1 : 25 000 (from 1:15 840)

Date of Photography: 1946

Site Number:	77, 78 & 79	**Wildfire Date:**	1966
Grid Reference:	E472440 N⁵257450 E472400 N⁵257560 E472490 N⁵257590	**Reference:**	Forestry Commission (1970)
District:	Norfolk	**Comments:**	
Locality:	Ted Ransleys Road (Styx Valley)		





Map Name:	Skeleton	**Map Name:**	Styx
Map Number:	4625	**Map Number:**	81/2 & 81/6
Scale:	1 : 25 000	**Scale:**	1 : 25 000 (from 1:15 840)
Date of Photography:	1984	**Date of Photography:**	1947

Site Number: 97

Grid Reference: E463040 N[5]269970

District: Norfolk

Locality: Chrisps Road (Florentine Valley)

Wildfire Date: 1966

Reference: M. White (pers. comm.)

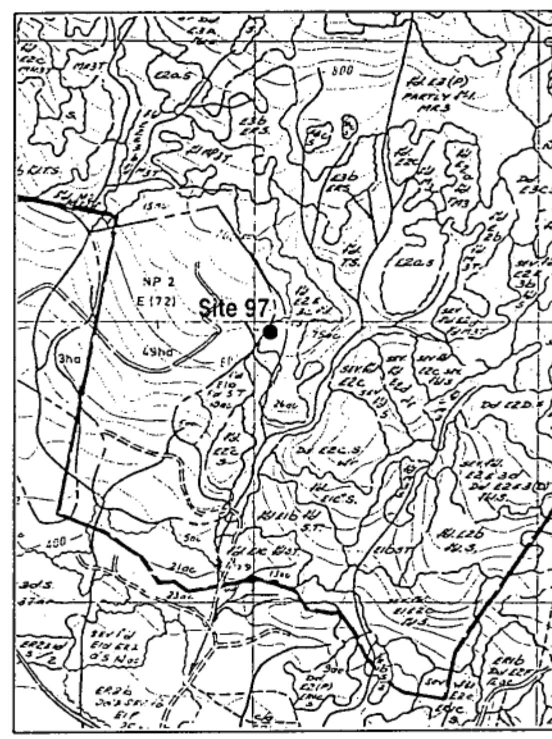
Comments:

Map Name: Aerial Photo Florentine-Maydena

Map Number: 1168-43 M931 Run 69

Scale: 1 : 20 000

Date of Photography: 1991



Map Name: Ellendale

Map Number: 74/13

Scale: 1:25 000 (from 1:15 840)

Date of Photography: 1967

Appendix 4. Site facing sheet and plot tally sheet.

		SITE	FACING SHEET			
Mixfor site no.						
site type		1.silv.regen (artificial)				
		2.silv.regen (natural)				
		3.wildfire				
		oldgrowth				
CFI plot no						
Current PI type				mmt date		by
Former PI type				map name		
old mapname				grid ref	E	
					N	
rock type						
altitude		m				
slope class(deg)						
aspect	(deg)					
topo		flat/gully/side-slope/ridge				
logging intensity		coupe yield /ha		Regen.	Wildfire	
1.clearfell-		s'log		burn	intensity	
2.partial-		p'wood		1.hot	1.crown	
3.nil-				2.cool	2.ground	
					3.peat	
					4.nil	
Site MDH						
species	height					
transect layout		direction		6 5		
		long axis		7 4		
				8 3		
				9 2		
				10 1		
Structural formula			T			
"walkabout notes"						

				PLOT TALLY SHEET							PLOT NO
cover class											
1	,<1%										
2	1-5%										
3	5-25%										
4	25-50%										
5	50-75%					diam	height	sprout	No. of		Stag
6	75-100%		EP no.	cover	density	seedling	seedling	code	stags		diameter
R/f	Acac	mela	2								
trees	Athe	mosc	22								
	Eucr	lucid	86								
	Noth	cunn	149								
	Phyl	aspl	166								
Other	Acac	deal	1								
trees	Acac	mucr	3								
	Acac	rice	5								
	Acac	vert	4								
	Euca	broo	246								
	Euca	dele	78								
	Euca	glob	229								
	Euca	niti	81								
	Euca	obli	82								
	Euca	ovat	83								
	Euca	regn	84								
	Euca	umi	242								
	Lept	lani	127								
	Lept	scop	131								
	Mela	squa	139								
Tall shrubs											
	Anodo	bigl	9								
	Anop	glan	10								
	Bank	marg	230								
	Bedf	sala	30								
	Cass	acul	47								
	Cena	niti	49								
	Heli	dend	238								
	Mono	glau	145								
	Noto	ligu	148								
	Olea	argo	151				Ground cover				
	Pheb	squa	165				Acae	nova	232		
	Pitt	bico	171				Aste	alpi	21		
	Pros	lasi	181				Care	sp.	244		
	Poma	apet	179				Clem	aris	51		
	Tasm	lanc	202				Cory	dila	234		
	Telo	trun	219				Dian	tasm	64		
	Zier	arbo	217				Drym	cyan	71		
Low shrubs							Gahn	gran	88		
	Aris	pedu	15				Hydr	hirt	233		
	Baue	rubi	28				Lepi	elat	124		

	Bill	long	32				Sarc	aust	236		
	Copr	niti	53				Unci	sp.	245		
	Copr	quad	54				Viol	hede	216		
	Corr	lawr	55								
	Cyat	glau	59			Ferns					
	Cyat	juni	61				Aspl	flac	17		
	Cyat	parv	62				Blec	fluv	35		
	Gaul	hisp	90				Blech	watt	40		
	Olea	lanc	247				Blech	nudu	37		
	Olea	lira	239				Cten	hete	57		
	Olea	pers	152				Cyat	aust	237		
	Olea	stel	240				Dick	anta	65		
	Orit	dive	156				Glei	micr	94		
	Pime	cine	167				Gram	bill	97		
	Pime	drup	168				Hist	inci	104		
	Prio	ceri	180				Hypo	rugo	114		
	Pseu	gunn	182				Hyme	aust	108		
	Pult	juni	231				Hyme	cupr	109		
	Rich	drac	241				Hyme	flab	110		
	Sene	line	243				Hyme	marg	111		
	Sene	sp.	235				Hyme	pelt	112		
	Troc	cunn	207				Hyme	raru	113		
	Troc	dist	208				Micr	dive	143		
	Troc	gunn	209				Poly	prol	177		
							Poly	veno	176		
							Pter	escu	183		
							Rumo	adia	195		
							Tmes	bill	204		

Appendix 5. Key to abbreviated species names.

Trees	
Acac deal	*Acacia dealbata*
Acac mela	*Acacia melanoxylon*
Acac mucr	*Acacia mucronata*
Acac vert	*Acacia verticillata*
Acac rice	*Acacia riceana*
Athe mosc	*Atherosperma moschatum*
Euca broo	*Eucalyptus brookeriana*
Euca dele	*Eucalyptus delegatensis*
Euca obli	*Eucalyptus obliqua*
Euca regn	*Eucalyptus regnans*
Eucr luci	*Eucryphia lucida*
Lept scop	*Leptospermum scoparium*
Mela squa	*Melaleuca squarrosa*
Noth cunn	*Nothofagus cunninghamii*
Phyl aspl	*Phyllocladus aspleniifolius*
Tall shrubs	
Anod bigl	*Anodopetalum biglandulosum*
Anop glan	*Anopterus glandulosus*
Cena niti	*Cenarrhenes nitida*
Mono glau	*Monotoca glauca*
Note ligu	*Notelaea ligustrina*
Olea argo	*Olearia argophylla*
Pheb squa	*Phebalium squameum*
Pitt bico	*Pittosporum bicolor*
Poma apet	*Pomaderris apetala*
Pros lasi	*Prostanthera lasianthos*
Tasm lanc	*Tasmannia lanceolata*
Zier arbo	*Zieria arborescens*
Low shrubs	
Aris pedu	*Aristotelia peduncularis*
Copr quad	*Coprosma quadrifida*
Cyat glau	*Cyathodes glauca*
Cyat juni	*Cyathodes juniperina*
Pime cine	*Pimelea cinerea*
Pime drup	*Pimelea drupacea*
Troc cunn	*Trochocarpa cunninghamii*
Troc dist	*Trochocarpa disticha*
Herbs and sedges	
Dian tasm	*Dianella tasmanica*
Gahn gran	*Gahnia grandis*
Hydr spp.	*Hydrocotyle* spp.
Lepi elat	*Lepidosperma elatius*

Epiphytic ferns	
Gram bill	*Grammitis billardierei*
Hyme aust	*Hymenophyllum australe*
Hyme cupr	*H. cupressiforme*
Hyme flab	*H. flabellatum*
Hyme pelt	*H. peltatum*
Hyme raru	*H. rarum*
Micr dive	*Microsorum diversifolium*
Rumo adia	*Rumohra adiantiformis*
Tmes bill	*Tmesipteris billardierei*
Ground ferns	
Blec nudu	*Blechnum nudum*
Blec watt	*Blechnum wattsii*
Dick anta	*Dicksonia antarctica*
Hist inci	*Histiopteris incisa*
Hypo rugo	*Hypolepis rugosula*
Poly prol	*Polystichum proliferum*
Pter escu	*Pteridium esculentum*
Climbers	
Bill long	*Billardiera longiflora*
Clem aris	*Clematis aristata*
Prio ceri	*Prionotes cerinthoides*

Appendix 6. Germination results from seed trays. (The frequency of all standing woody species recorded at each site is also shown to compare the composition of the seed bank with the extant vegetation.)

Site 43

Woody species	Frequency %	Germinants March 1992	Germinants March 1993
Acacia melanoxylon	10	10	1
Atherosperma moschatum	10		
Eucalyptus obliqua	50		
Eucryphia lucida	70		
Nothofagus cunninghamii	90		
Phebalium squameum	-	0	5
Pittosporum bicolor	20		
Pomaderris apetala	-	13	16

Non-woody species

March 1992: *Epilobium* sp.[1], *Gahnia grandis*

March 1993: *Epilobium* sp.[1], *Gahnia grandis, Isolepis* sp., *Sagina procumbens*

[1] = non-native species

Site 46

Woody species	Frequency %	Germinants March 1992	Germinants March 1993
Aristotelia peduncularis	10		
Cenarrhenes nitida	10		
Coprosma quadrifida	20		
Correa lawrenciana	70		
Cyathodes juniperina	20		4
Eucalyptus obliqua	10		
Eucryphia lucida	50		
Nothofagus cunninghamii	100		
Phyllocladus aspleniifolius	10		
Trochocarpa cunninghamii	70		

<u>Non-woody species</u>

March 1992: *Epilobium* sp.[1], *Solanum* sp.[1]

March 1993: *Epilobium* sp.[1], *Gnaphalium* sp., *Sagina procumbens, Schoenus apogon*

[1] = non-native species

Site 47

Woody species	Frequency %	Germinants March 1992	Germinants March 1993
Atherosperma moschatum	60		
Coprosma quadrifida	20		
Eucalyptus obliqua	10		
Nothofagus cunninghamii	100		
Olearia argophylla	40		
Pomaderris apetala	-	1	

Non-woody species

March 1992: *Epilobium* sp.[1], *Senecio linearifolius*

March 1993: *Epilobium* sp.[1], *Isolepis*, *Sagina procumbens*

[1] = non-native species

Site 49

Woody species	Frequency %	Germinants March 1992	Germinants March 1993
Acacia sp	-	1	
Atherosperma moschatum	10		
Eucalyptus obliqua	80		
Monotoca glauca	20		9
Nothofagus cunninghamii	100		
Pittosporum bicolor	10		
Sprengelia incarnata	-		1

Non-woody species

March 1992: *Epilobium* sp.[1], *Isolepis* sp., *Juncus* sp.

March 1993: *Epilobium* sp.[1], *Gahnia grandis*, *Gnaphalium* sp., *Isolepis* sp.

[1] = non-native species

Site 63

Woody species	Frequency %	Germinants March 1992	Germinants March 1993
Anodopetalum biglandulosum	100		
Anopteris glandulosus	80		
Atherosperma moschatum	50		
Cenarrhenes nitida	70		
Coprosma nitida	10		
Cyathodes glauca	20		
Cyathodes juniperina	10		
Eucalyptus obliqua	40		
Eucryphia lucida	30		
Gaultheria hispida	-		2
Nothofagus cunninghamii	90		
Olearia persooniodes	30		
Orites diversifolia	50		
Phyllocladus aspleniifolius	90	2	2
Pittosporum bicolor	40		
Tasmannia lanceolata	30		
Trochocarpa disticha	20		
Trochocarpa gunnii	20		12

Non-woody species

March 1992: *Acer* sp.[1], *Clematis vitalba*[1], *Epilobium* sp.[1], *Gahnia grandis, Gnaphalium* sp., *Hydrocotyle* sp.,

March 1993: *Epilobium* sp.[1], *Gnaphalium* sp., *Hydrocotyle* sp., *Sagina procumbens, Schoenus apogon, Solanum* sp.

[1] = non-native species

Site 64

Woody species	Frequency %	Germinants March 1992	Germinants March 1993
Acacia dealbata	-	8	
Acacia melanoxylon	-	1	
Aristotelia peduncularis	30		
Atherosperma moschatum	100		
Cassinia aculeata	-	6	
Coprosma quadrifida	60		
Eucalyptus regnans	70		
Gaultheria hispida	10		
Olearia argophylla	20		
Pittosporum bicolor	10		
Tasmannia lanceolata	30		

Non-woody species

March 1992: *Clematis vitalba*[1], *Epilobium* sp.[1] , *Gahnia grandis, Gnaphalium* sp., *Juncus* sp., *Sagina procumbens, Schoenus apogon, Senecio linearifolius*

March 1993: *Epilobium* sp.[1], *Gahnia grandis, Gnaphalium* sp., *Isolepis* sp., *Juncus* sp., *Sagina procumbens, Schoenus apogon, Senecio linearifolius*

[1] = non-native species

Site 65

Woody species	Frequency %	Germinants March 1992	Germinants March 1993
Acacia sp.	-		1
Anodopetalum biglandulosum	100		
Anopterus glandulosus	50		
Atherosperma moschatum	100		
Coprosma quadrifida	20		
Cyathodes juniperina	-		1
Eucalyptus obliqua	30		
Eucryphia lucida	90	3	
Nothofagus cunninghamii	30		
Phyllocladus aspleniifolius	10	1	
Pittosporum bicolor	20		
Tasmannia lanceolata	10		

Non-woody species

March 1992: *Epilobium* sp.[1]

March 1993: *Epilobium*[1], *Gahnia grandis, Gnaphalium* sp., *Isolepis* sp., *Sagina procumbens, Schoenus apogon*

[1] = non-native species

Site 69

Woody species	Frequency %	Germinants March 1992	Germinants March 1993
Acacia sp.	-		1
Cenarrhenes nitida	60		
Cyathodes juniperina	70		24
Eucalyptus nitida	20		
Monotoca glauca	40		2
Notelaea ligustrina	20		
Nothofagus cunninghamii	100		
Phebalium squameum	-		3
Phyllocladus aspleniifolius	40	1	
Tasmannia lanceolata	70		
Trochocarpa cunninghamii	40		

Non-woody species

March 1992: *Epilobium* sp.[1]

March 1993: *Epilobium* sp.[1], *Isolepis* sp., *Sagina procumbens,*

[1] = non-native species

Site 83

Woody species	Frequency %	Germinants March 1992	Germinants March 1993
Anodopetalum biglandulosum	90		
Atherosperma moschatum	70		
Eucalyptus regnans	50		
Nothofagus cunninghamii	90		
Phyllocladus aspleniifolius	10	1	
Trochocarpa gunnii	-		2

Non-woody species

March 1992: *Epilobium* sp.[1], *Isolepis* sp.

March 1993: *Epilobium* sp.[1], *Isolepis* sp.

[1] = non-native species

Appendix 7. Summary of information for ring-counted stems. (LAG = the stand age - the number of rings).

Site/plot	Site type	Stand age	Species	Stump ht m	Stump diameter cm	Rings	LAG	Origin
38/10	wildfire	28	*A. mos*	0.0	0.9	11	17	seed
78/2	wildfire	25	*A. mos*	0.0	0.3	8	17	seed
85/1	logged	25	*A. mos*	0.0	0.5	10	15	seed
97/2	wildfire	25	*A. mos*	0.0	0.4	6	19	seed
53/9	logged	23	*A. mos*	0.1	2.4	11	12	seed
59/1	logged	30	*A. mos*	0.1	3.1	24	6	seed
70/9	wildfire	25	*A. mos*	0.1	7.8	25	0	sprout
75/5	wildfire	25	*A. mos*	0.1	6.2	25	0	sprout
77/7	wildfire	25	*A. mos*	0.1	1.5	15	10	seed
79/2	wildfire	25	*A. mos*	0.1	1.5	10	15	seed
92/9	logged	30	*A. mos*	0.1	4.9	24	6	seed
81/6	logged	28	*A. mos*	0.2	11.2	27	1	sprout
76/6	wildfire	25	*A. mos*	0.3	13.9	23	2	sprout
36/2	wildfire	30	*E. luc*	0.0	1.0	30	0	seed
40/8	wildfire	30	*E. luc*	0.1	6.1	21	9	seed
55/5	logged	29	*E. luc*	0.1	5.3	19	10	seed
58/8	wildfire	25	*E. luc*	0.1	6.6	24	1	seed
60/5	logged	21	*E. luc*	0.1	1.8	17	4	seed
70/10	wildfire	25	*E. luc*	0.1	7.5	22	3	sprout
89/1	logged	30	*N. cun*	0.0	1.4	22	8	seed
79/2	wildfire	25	*N. cun*	0.0	0.5	12	13	seed
97/2	wildfire	25	*N. cun*	0.0	0.8	14	11	seed
38/5	wildfire	28	*N. cun*	0.1	1.6	19	9	seed
53/10	logged	23	*N. cun*	0.1	2.8	16	7	seed
36/5	wildfire	30	*N. cun*	0.1	1.5	16	14	seed
57/3	wildfire	25	*N. cun*	0.1	5.7	16	9	seed
58/7	wildfire	25	*N. cun*	0.1	9.7	21	4	seed
75/8	wildfire	25	*N. cun*	0.1	3.5	23	2	sprout
77/2	wildfire	25	*N. cun*	0.1	3.7	25	0	sprout
78/7	wildfire	25	*N. cun*	0.1	3.7	21	4	seed
87/4	logged	30	*N. cun*	0.1	2.9	23	7	seed
40/7	wildfire	30	*N. cun*	0.2	14.4	30	0	sprout
56/1	logged	27	*N. cun*	0.2	10.8	26	1	sprout
59/7	logged	30	*N. cun*	0.2	12.0	23	7	sprout
67/5	logged	21	*N. cun*	0.2	13.8	21	0	sprout
81/4	logged	28	*N. cun*	0.2	15.5	24	4	seed
85/5	logged	25	*N. cun*	0.2	8.3	22	3	seed

Site/plot	Site type	Stand age	Species	Stump ht m	Stump diameter cm	Rings	LAG	Origin
86/9	logged	30	*N. cun*	0.2	10.3	21	9	seed
92/9	logged	30	*N. cun*	0.2	7.2	24	6	seed
38/10	wildfire	28	*P. asp*	0.0	1.0	9	19	seed
40/9	wildfire	30	*P. asp*	0.0	1.8	16	14	seed
70/6	wildfire	25	*P. asp*	0.0	1.2	17	8	seed
75/2	wildfire	25	*P. asp*	0.0	0.6	20	5	seed
78/6	wildfire	25	*P. asp*	0.1	3.9	19	6	seed
84/1	logged	25	*P. asp*	0.1	5.3	23	2	seed
53/5	logged	23	*P. asp*	0.1	2.7	20	3	seed
55/2	logged	29	*P. asp*	0.1	6.3	21	8	seed
77/2	wildfire	25	*P. asp*	0.1	4.9	18	7	seed
79/2	wildfire	25	*P. asp*	0.1	2.6	16	9	seed